



# **A Natural Resource Condition Assessment for Sequoia and Kings Canyon National Parks**

## *Appendix 15b - Animals of Conservation Concern, Supplemental Information*

Natural Resource Report NPS/SEKI/ NRR—2013/665.15b



**ON THE COVER**

Giant Forest, Sequoia National Park  
Photography by: Brent Paull

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# Contents

	Page
Assessments .....	1
A. Threatened and Endangered Species .....	1
A1. The American Pika - <i>Ochotona princeps</i> .....	1
Abstract .....	1
Description and taxonomy .....	1
Life History .....	2
Distribution .....	3
Conservation Status .....	4
Threats and trends .....	6
Conservation in Sequoia and Kings Canyon .....	7
Data Gaps & Research Priorities .....	7
Summary and Recommendations .....	8
Data Sources .....	8
A.2. Sierra Nevada Red Fox – <i>Vulpes vulpes necator</i> .....	9
Abstract .....	9
Background and Taxonomy .....	10
Physical Description .....	10
Taxonomy and Genetics .....	11
Life History .....	11
Distribution .....	13
Status .....	17
Threats .....	17
Data Gaps, Research Priorities, and Conservation .....	18

## Contents (continued)

	Page
Summary and Recommendations .....	18
Data Sources .....	19
A3. Wolverine – <i>Gulo gulo</i> .....	20
Abstract .....	20
Description and taxonomy .....	20
Life History .....	21
Distribution .....	21
Conservation Status .....	23
Threats and trends .....	23
Conservation in Sequoia – Kings Canyon .....	24
Data Gaps & Research Priorities .....	24
Summary and Recommendations .....	25
Data Sources .....	25
A4. California Condor ( <i>Gymnogyps californianus</i> ) .....	26
Abstract .....	26
Species Background.....	26
Distribution and population trends.....	27
Conservation concern.....	28
Continued stewardship.....	28
A5. Brown Bear ( <i>Ursus arctos</i> ).....	29
Abstract .....	29
Species Background.....	29
Distribution and Population Trends .....	30

## Contents (continued)

	Page
Conservation Status .....	31
Management potential in Sequoia Kings Canyon National Park.....	31
A.6. Sierra Nevada Bighorn Sheep - <i>Ovis canadensis sierrae</i> .....	33
Abstract .....	33
Species background .....	33
Distribution .....	35
Condition and Trends.....	37
Threats.....	40
Management.....	41
Data Gaps & Research Priorities .....	43
Summary and Recommendations .....	43
Data Sources .....	44
A.7. The Fisher – <i>Martes pennanti</i> .....	45
Abstract .....	45
Background and taxonomy .....	45
Life History.....	46
Distribution .....	48
Conservation Status .....	50
Threats.....	50
Data Gaps and Research Needs in Sequoia-Kings Canyon .....	52
Summary and Recommendations .....	53
Data Sources .....	54
A.8. Mountain Yellow-legged Frogs: <i>Rana muscosa</i> and <i>Rana sierrae</i> .....	55

## Contents (continued)

	Page
Abstract .....	55
Species Background/Life History .....	56
Global and Regional Context/Population Genetics .....	57
Population Threats .....	58
Past and Present Management .....	61
Future Action .....	63
B. Park Limited Taxa.....	64
B1. Slender Salamanders: <i>Batrachoseps kawia</i> and <i>Batrachoseps regius</i> .....	64
Abstract .....	64
Species Background/Life History .....	64
Global and Regional Context.....	65
Distribution .....	65
Population Biology .....	66
Threats.....	67
Future Action .....	67
B2. Sierra Nevada Chipmunks— <i>Tamias</i> spp. ....	68
Abstract .....	68
Taxonomy and General Background .....	68
The Grinnell Resurvey Project.....	68
Sequoia and Kings Canyon National Parks .....	69
C. Sensitive Fishes, Native and Introduced. ....	71
Abstract.....	71
Native taxa .....	71



## Contents (continued)

	Page
Non-native taxa.....	71
Salmonid Taxonomy.....	71
Salmonid Life History .....	72
C1. Native Trout .....	73
The Golden Trout Complex, <i>Oncorhynchus mykiss</i> . ....	73
Little Kern Golden Trout. ( <i>O. mykiss whitei</i> ) .....	73
Kern River rainbow trout ( <i>O. mykiss gilberti</i> ).....	75
C2. Trout Introduced to Sequoia and Kings Canyon National Parks .....	78
Global Distribution .....	78
Moving from Current Condition to Stewardship .....	86
D. Invasive Species.....	93
D.1. New Zealand Mudsnail <i>Potamopyrgus antipodarum</i> .....	93
Abstract .....	93
Species background .....	93
Regional Context .....	93
Distribution in Sequoia and Kings Canyon National Parks.....	94
History of spread, introduction mechanisms .....	95
Description of threats from <i>P. antipodarum</i> .....	97
Management.....	98
Summary and Management Recommendations.....	98
D.2. Brown-headed Cowbird ( <i>Molothrus ater</i> ) .....	99
Abstract .....	99
Species Background.....	99

## Contents (continued)

	Page
Distribution and population trends.....	99
Conservation concern.....	102
Management.....	103
D.3. Feral Pigs ( <i>Sus scrofa</i> ) .....	104
Abstract .....	104
Background and taxonomy .....	104
Life History .....	104
Distribution and population trends.....	105
Conservation concern.....	106
Management.....	106
D.4. Cattle Trespass .....	107
Abstract .....	107
Background .....	107
Regional Impacts .....	108
Sequoia and Kings Canyon.....	110
Management.....	111
Cattle Trespass Stewardship .....	113
Summary/Recommendations .....	113
D.5. Bullfrog <i>Rana catesbeiana</i> .....	115
Abstract .....	115
Species background .....	115
Distribution .....	115
Sequoia and Kings Canyon National Parks distribution.....	116

## Contents (continued)

	Page
Threats.....	117
Data needs .....	118
Summary / recommendations .....	118
Literature Cited .....	119



# Figures

	Page
<b>Figure A1.1:</b> American Pika ( <i>Ochotona princeps</i> ) .....	1
<b>Figure A1.2:</b> Occurrence of five pika subspecies across the California Sierra Nevada and their relation to county boundaries and the Sequoia and Kings Canyon National Parks.....	3
<b>Figure A1.2:</b> American Pika Global distribution.....	3
<b>Figure A1.3.</b> Pika observations within Sequoia and Kings Canyon National Parks .....	5
<b>Figure A2.1.</b> Recent photo of a cross-phase Sierra Nevada red fox taken at a remote camera station in 2010 near Sonora Pass, California (courtesy of S. Lisius, US Forest Service). .....	9
<b>Figure A2.2.</b> Radio-collared Sierra Nevada red fox .....	10
<b>Figure A2.3.</b> Photo of a Sierra Nevada red fox at a baited camera station near Sonora Pass, California in 2010 (courtesy of S. Lisius, US Forest Service).....	12
<b>Figure A2.4.</b> Distribution of the red fox ( <i>Vulpes vulpes</i> ) in North America (NatureServe 2007). .....	13
<b>Figure A2.5.</b> Distribution of the twelve subspecies of red fox ( <i>Vulpes vulpes</i> ) in North America as presented by Ables (1975).....	14
<b>Figure A2.6.</b> Historical distribution of the Sierra Nevada red fox ( <i>Vulpes vulpes necator</i> ) in California (from Grinnell et al. 1937) .....	16
<b>Figure A2.7.</b> Distribution of unconfirmed reports of Sierra Nevada red fox from the parks' observation database and location of sites surveyed for carnivores (2002-2004) with no red fox detections in Sequoia-Kings Canyon National Parks.....	16
<b>Figure A3.1</b> Photo of wolverine ( <a href="http://en.wikipedia.org/wiki/File:Wolverine_on_rock.jpg">http://en.wikipedia.org/wiki/File:Wolverine_on_rock.jpg</a> ).....	20
<b>Figure A3.2.</b> Current known and possible distribution of wolverines in North America (Copeland and Whitman 2003) .....	22
<b>Figure A3.3.</b> Reports of wolverine in SEKI from 1903-2008 (SEKI 2010).....	23
<b>Figure A4.1.</b> Historic and current California Condor distribution (Snyder and Schmitt 2002) .....	27
<b>Figure A4.2.</b> California Condor range in the 20 <sup>th</sup> century. (Ventana Wildlife Society).....	27

## Figures (continued)

	Page
<b>Figure A5.1</b> Current global distribution of <i>Ursus arctus</i> separated by subspecies.....	32
<b>Figure A5.2</b> Former and present distribution of <i>Ursus arctos</i> in North America.....	32
<b>Figure A6.1.</b> Distribution of all subspecies of <i>Ovis canadensis</i> .....	35
<b>Figure A6.2.</b> Depiction of the California Natural Diversity Database (CNDDB) distribution data of the three bighorn sheep subspecies found within California.....	36
<b>Figure A6.3.</b> Sequoia and Kings Canyon observations of bighorn sheep between 1935 and 2009 .....	38
<b>Figure A6.4.</b> Recent reproductive base histories for 6 herd units of bighorn sheep in the Sierra Nevada for which adequate data exist .....	39
<b>Figure A6.5.</b> Units proposed by the U.S. Fish and Wildlife Service as critical habitat for Sierra Nevada Bighorn Sheep. Figure from CDFG 2008a.....	43
<b>Figure A7.1.</b> Photograph of a fisher at a remote camera station near Grant Grove, Kings Canyon National Park. ....	46
<b>Figure A7.2.</b> Current global distribution of the fisher (NatureServe 2010).....	49
<b>Figure A7.3.</b> Distribution and conservation status of the west coast fisher population (NatureServe 2010). ....	49
<b>Figure A7.4.</b> Comparison of historical and current records of fisher in the Sierra Nevada (figure from Zielinski et al. 2004). ....	51
<b>Figure A7.5.</b> Distribution of fisher observations from database and verified fisher detections from surveys conducted from 2002-2004.....	51
<b>Figure A8.1:</b> Map adapted from Vredenburg et al 2007. Points represent localities of genetic samples of mountain yellow-legged frogs.....	57
<b>Figure 8.2:</b> Foodweb diagram contrasting energy flow pathways to consumers under (a) historical, fishless conditions and (b) as influenced by introduced trout .....	59
<b>Figure B1.1.</b> Locations of <i>Batrachoseps</i> in and around the park.....	66
<b>Figure C1.1.</b> The Little Kern River and its tributaries, with Little Kern golden trout critical habitat shaded in blue. SEKI lies north of the gray border, indicating the small amount of Little Kern golden trout habitat within the park boundary. ....	74

## Figures (continued)

	Page
<b>Figure C1.2.</b> The Kern River basin within the SEKI boundary, with major tributaries shown .....	76
<b>Figure C1.3.</b> Worldwide distribution of <i>Oncorhynchus mykiss</i> .....	78
<b>Figure C1.4.</b> Worldwide distribution of <i>Salmo trutta</i> .....	79
<b>Figure C1.5.</b> Adapted from Moyle et al (1996) .....	80
<b>Figure C1.6.</b> Distribution of rainbow, brown and brook trout in California .....	80
<b>Figure C1.7.</b> Distribution of known non-native trout presence in SEKI at high elevations (unpublished data, Knapp, RA.) .....	82
<b>Figure C1.8.</b> Distribution of known trout presence in SEKI at low and mid elevations .....	84
<b>Figure D1.1.</b> Current United States distribution of <i>Potamopyrgus antipodarum</i> .....	94
<b>Figure D1.2.</b> Maps showing the invasion of <i>Potamopyrgus antipodarum</i> in the Western US. Data are from <a href="http://esg.montana.edu">http://esg.montana.edu</a> . .....	95
<b>Figure D1.3.</b> Point locations of <i>P. antipodarum</i> in region surrounding Sequoia and Kings Canyon National Parks .....	96
<b>Figure D2.1.</b> Brown-headed Cowbird distribution (Lowther 1993) .....	100
<b>Figure D2.2.</b> Recorded observations of Brown-headed Cowbirds in Sequoia and Kings Canyon (SEKI 2010). .....	101
<b>Figure D2.3:</b> Map of Brown-headed Cowbird observations in Sequoia and Kings Canyon (SEKI 2010).....	101
<b>Figure D3.1</b> Map of the current distribution of feral pigs in the United States, and specifically in California.....	105
<b>Figure D4. 1.</b> The distribution of preferred forage vegetation (grasses and sedges) for cattle and locations of known cattle trespass within Sequoia and Kings Canyon National Parks. ....	111
<b>Figure D4.2.</b> The ownership and management status of lands bordering Sequoia-Kings Canyon National Park and the known points of cattle trespass .....	112
<b>Figure D5.1</b> Distribution of <i>R. catesbeiana</i> in the United States.....	116

## Figures (continued)

	Page
<b>Figure D5.2</b> Confirmed presence of <i>R. catesbeiana</i> in Europe. ....	116
<b>Figure D5.3.</b> Location of the known bullfrog population in Sequoia and Kings Canyon National Parks. ....	117



## Tables

	Page
<b>Table A1.1:</b> Subspecies natural heritage records by California county .....	4
<b>Table A1.2:</b> NatureServe and IUCN conservation rankings of the five California pika subspecies .....	5
<b>Table A3.1.</b> Wolverine observations in SEKI from 1925-1948 (SEKI 2010) .....	22
<b>Table A6.1.</b> Summary of presence of the three bighorn sheep subspecies found in California counties. ....	36
<b>Table A6.2.</b> Summary of population data as of summer 2008.....	37
<b>Table B2.1.</b> Chipmunk species of Sequoia and Kings Canyon National Parks. ....	68
<b>Table B2.2.</b> Elevational range shifts observed along the Grinnell Yosemite resurvey transect for four chipmunk species .....	69
Table D1.1. <i>Potantopyrgus antipodarum</i> observations near Sequoia and Kings Canyon National Parks .....	97
<b>Table D2.1.</b> Brown-headed Cowbird detections and estimated densities by habitat type.....	102



## Assessments

### A. Threatened and Endangered Species

#### A1. The American Pika - *Ochotona princeps*

##### **Abstract**

Two subspecies of the American Pika (*O. p. albata* and *O. p. muiri*, referred to hereafter, collectively as ‘pika’) inhabit talus slopes in the alpine zone of Sequoia and Kings Canyon National Parks. The species appears to be widespread among these specialized habitats within the parks and the southern Sierra Nevada range. Due to their remote location, few human threats exist for the pika. The greatest threat to this species is likely to be climate change. The pika’s relatively limited temperature tolerance makes them vulnerable to increasingly warm temperatures.

- **Species global status:** G5 – Common (Widespread and abundant)  
*O. p. albata*: T3 – Vulnerable (Moderate risk of extinction or elimination)  
*O. p. muiri*: T3 – Vulnerable
- **Park status:** Patchy - pika are frequently found within specialized habitat of the alpine zone.
- **Overall integrity:** High - While populations elsewhere (i.e. the Great Basin) have experienced recent declines, pika appear to be widespread within the alpine zone of the Southern Sierra Nevada and Sequoia and Kings Canyon National parks.
- **Certainty of integrity:** Moderate - More systematic surveys of pika habitat are needed.
- **Metric to evaluate integrity:** Percent occupancy of optimal-appearing habitat in the alpine zone (88% in the Sierra Nevada, Millar and Westerfall 2010)

##### **Description and taxonomy**

The American pika (*Ochotona princeps*) is a small rodent with short limbs, “buried” tail, and moderately large ears (Smith and Weston 1990). Many populations are insular, particularly those below timberline (Smith 1974a, Orr 1977). While they are often found above tree-line where edible grasses and forbs still occur, they can also be found in rocky areas at lower elevations (NatureServe 2009). However, the lower elevation limit of pika distribution may be set by excessive heat (MacArthur and Wang 1973, Smith 1974b).

The pika is in the family Ochotonidae and order Lagomorpha (Figure A1.1).



**Figure A1.1:** American Pika (*Ochotona princeps*).  
Photo from  
[http://en.wikipedia.org/wiki/American\\_Pika](http://en.wikipedia.org/wiki/American_Pika)

The family is composed of one genus, *Ochotona*, which encompasses 30 species. Five subspecies of American pika (*O. princeps*) occur only within California. These include the Mt. Whitney pika (*O. p. albata*), the Yosemite pika (*O. p. muiri*), the gray-headed pika (*O. p. schisticeps*), the White Mountains pika (*O. p. sheltoni*), and the Taylor pika (*O. p. taylori*) (NatureServe 2009). Note that the taxonomy used by the United States Fish and Wildlife Service lists one subspecies (*O.p. schisticeps*) as spanning the Sierra Nevada and Great Basin where NatureServe considers this distribution to be made up of a number of subspecies (FWS 2010).

### **Life History**

**Habitat.** The pika is an alpine habitat specialist and is generally associated with rocky talus slopes that occur at the talus-meadow interface (NatureServe 2009) and in rock-ice feature till (Millar and Westerfall 2010). Occasionally they will inhabit mine tailings or piles of lumber or scrap metal (NatureServe 2009). Pika co-occurs with yellow-bellied marmot (*Marmota flaviventris*), Belding's ground squirrel (*Uroditellus beldingi*), and various species of *Microtus spp.* (DFG 1999). Predators include weasels (*Mustela spp.*), American marten (*Martes americana*), coyote (*Canis latrans*) and raptors (Smith and Weston 1990, DFG 1999).

**Diet.** The pika forages on the ground surface, primarily within 6 m of rock or talus in alpine meadows and the edges of adjoining habitats such as alpine dwarf-shrub, subalpine conifer, lodgepole pine, montane chaparral, montane riparian, sage, pinyon-juniper, pine, mixed conifer and aspen habitat types (DFG 1999). It eats a wide variety of plants, including grasses, sedges, forbs, some flowering plants and shoots of woody vegetation (NatureServe 2009). Forbs and tall grasses are often cached, while short grasses are often eaten directly (Smith and Weston 1990). During winter, pikas forage in snow tunnels and will harvest cushion plants and lichen (Smith and Weston 1990). Water needs are met from available surface water and food (DFG 1999).

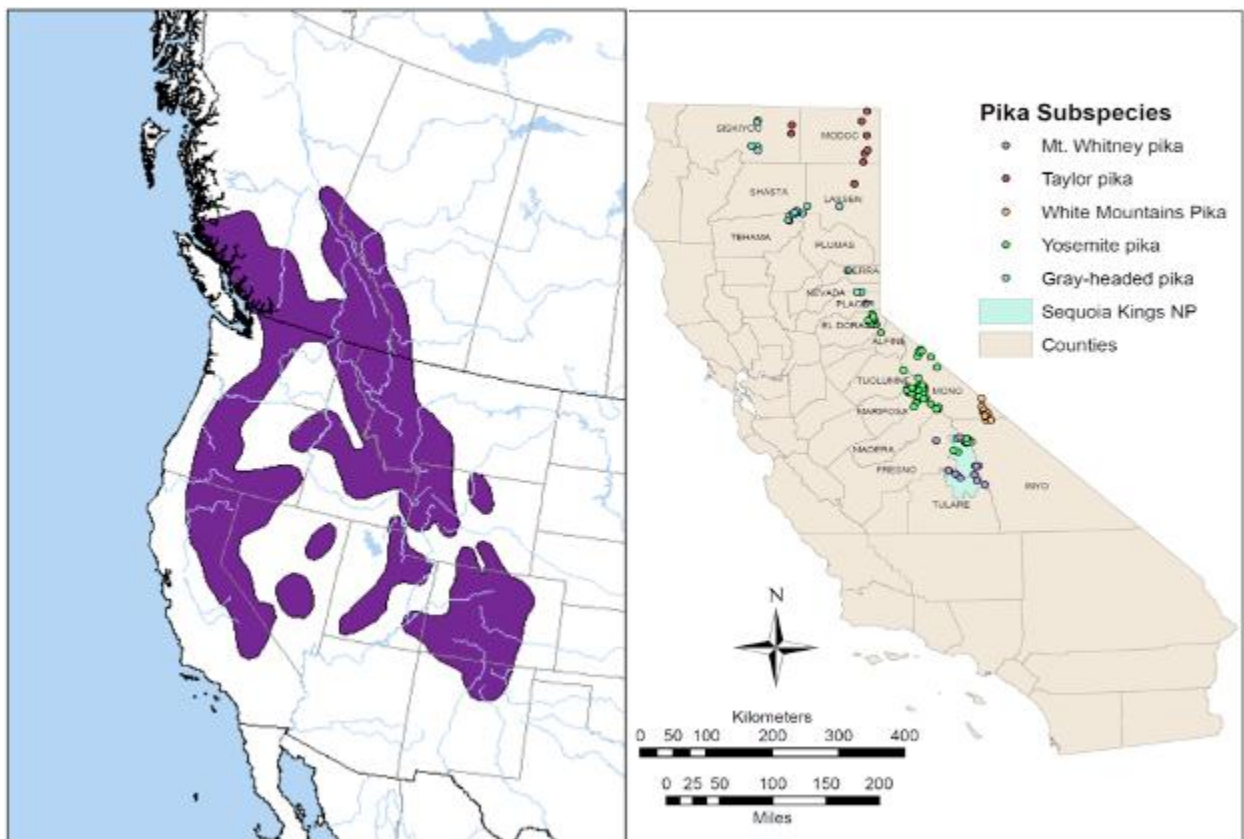
**Behavior.** Pikas exhibit diurnal activity throughout the year and show no seasonal movement or migration (DFG 1999, NatureServe 2009). Activity tends to be a function of altitude and temperature. At higher altitudes pikas forage for longer periods and farther from cover (Smith 1974b) and high temperatures can limit pika activity (MacArthur and Wang 1973, Smith 1974b, Smith and Weston 1990). Population densities vary from 4-22 individuals per ha (Millar 1974), and average home ranges are approximately 0.3 ha (Barash 1973, Kawamichi 1976). Juveniles tend to stay on natal home ranges or move to adjacent areas (NatureServe 2009).

**Reproduction and Survival.** Mating occurs in the spring and is highly dependent on snowmelt conditions with reproduction beginning later at higher altitudes (Smith and Weston 1990, DFG 1999). Litters are typically made up of 3 or 4 young, but range from 1 to 6 (DFG 1999) with an average of 3 at higher elevations (Smith and Weston 1990). Juvenile mortality is low (DFG 1999) and appears tied to availability of vacant territories (Smith and Weston 1990). Likelihood of dispersing across non-talus habitat is low (Smith and Weston 1990). Pikas are extremely sensitive to temperature; death can occur after brief exposures to ambient temperatures greater than 77.9 °F (FWS 2010).

### **Distribution**

**Global.** The American Pika (*Ochotona princeps*) is distributed across alpine and subalpine regions of the western United States and southwestern Canada. At the southernmost extent of its range, the occurrence of pikas increases progressively with elevation. States that support pika populations include California, Colorado, Idaho, Montana, New Mexico, Nevada, Oregon, Utah, Washington and Wyoming. Pikas are also found within the Canadian provinces of Alberta and British Columbia (Figure A1.2; NatureServe 2009).

**California.** Incomplete distribution data indicate that five subspecies of American Pika are native to the Sierra Nevada range in California. These subspecies include the Mt. Whitney pika (*O. p. albata*), the Yosemite pika (*O. p. muiri*), the gray-headed pika (*O. p. schisticeps*), the White Mountains pika (*O. p. sheltoni*) and the Taylor pika (*O. p. taylori*) (Figure A1.3; CNDDDB 2010).



**Figure A1.2:** American Pika Global distribution. Figure adapted from NatureServe <http://www.natureserve.org/explorer/index.htm> (Patterson et al. 2003).

**Figure A1.2:** Occurrence of five pika subspecies across the California Sierra Nevada and their relation to county boundaries and the Sequoia and Kings Canyon National Parks.

Natural heritage records of pika exist for 18 California counties. Many of these observations are quite old and it is believed that many populations may have since been extirpated (NatureServe 2009). However, recent observations recorded within Sequoia

and Kings Canyon National Parks and surveys completed by Millar and Westfall (2010) indicate that pika populations remain in at least those counties surveyed (Alpine, Fresno, Inyo, Mariposa, Mono, Tulare and Tuolumne). These observations suggest that the absence of recent natural heritage records does not necessarily suggest local extirpation and more formal surveys are needed to determine the current status of pika across California.

**Table A1.1:** Subspecies natural heritage records by California county. Indications of existing records and possible extirpations were acquired from the NatureServe species account (NatureServe 2009).

	Alpine	El Dorado	Fresno	Inyo	Lassen	Madera	Mariposa	Modoc	Mono	Nevada	Placer	Plumas	Shasta	Sierra	Siskiyou	Tehama	Tulare	Tuolumne
Mt. Whitney			•†	•					•†								•†	
Yosemite	•†	•†	•†	•†		•	•		•									•
Gray-headed		•†			•†					•†	•†	•†	•†	•†	•†	•†		
White Mountains				•†					•†									
Taylor					•†			•†							•†			

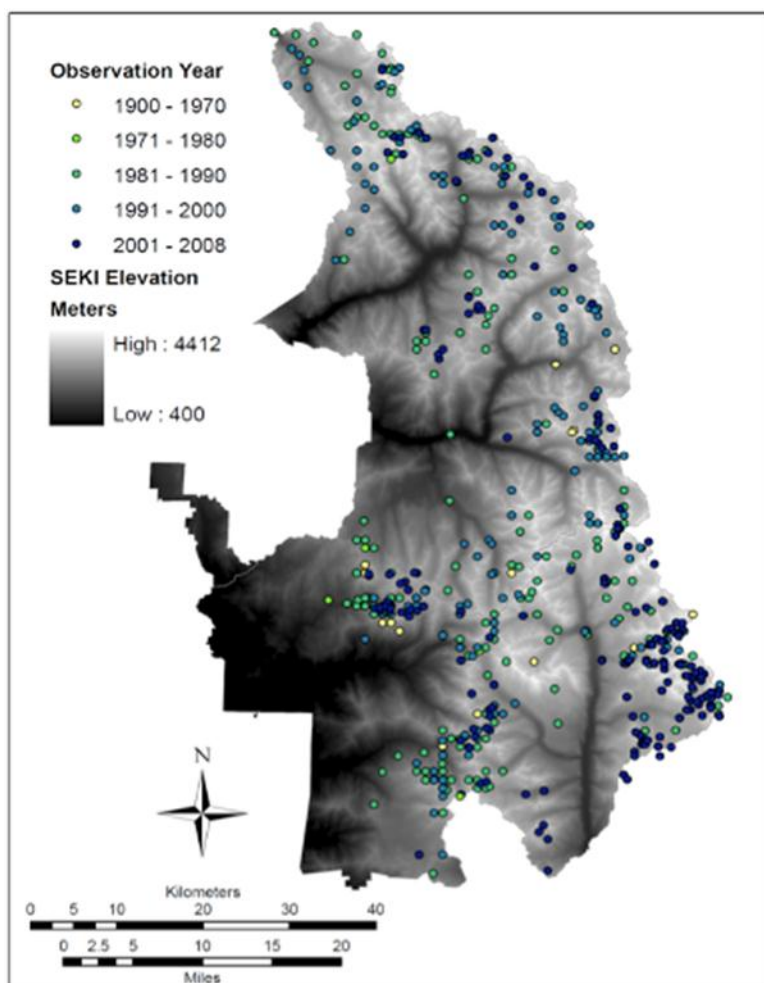
• Natural heritage records exist for county  
†Extirpated/possibly extirpated

### Sequoia and Kings Canyon National Parks

Two of the five pika subspecies that occur in California are present within Sequoia and Kings Canyon National Parks. These include Yosemite pika and Mt. Whitney pika, with Mt. Whitney pika found almost exclusively within the parks' boundaries. Recent observations indicate that pika populations remain widespread across areas of high elevation within park boundaries (Figure A1.3).

### **Conservation Status**

As a species, the American pika is considered secure by NatureServe (G5 ranking) and of least concern by the IUCN. However, of the five subspecies of American pika found in California, *O. p. sheltoni* (not located within Sequoia and Kings Canyon) is categorized as Critically Imperiled and the other four are categorized as Vulnerable (Table A1.2; NatureServe 2009, IUCN 2010).



**Figure A1.3.** Pika observations within Sequoia and Kings Canyon National Parks. Color categorization indicate date of observation and an altitude gradient illustrates the pika's preference for alpine habitats.

**Table A1.2:** NatureServe and IUCN conservation rankings of the five California pika subspecies. Note that there is a level of uncertainty surrounding the NatureServe T-rankings of these subspecies. For example, for *O. p. albata* the T-ranking is listed as a range from T2 – T4 with T3 as the rounded global status. The rounded global status of each subspecies is listed here.

Scientific Name	Common Name	NatureServe Ranking (Intraspecific Taxon)	IUCN Ranking
<i>O. p. albata</i>	Mt. Whitney pika	Vulnerable – T3	Not listed as threatened
<i>O. p. muiri</i>	Yosemite pika	Vulnerable – T3	Not listed as threatened
<i>O. p. schisticeps</i>	gray-headed pika	Vulnerable – T3	Not listed as threatened
<i>O. p. sheltoni</i>	White Mountains pika	Critically Imperiled – T1	Vulnerable – D1
<i>O. p. taylori</i>	Taylor pika	Vulnerable – T3	Not listed as threatened

### ***Threats and trends***

Owing to concern over possible threats to pika and the possible need for listing, The US Fish and Wildlife Service analyzed potential factors that could affect the habitat or range of the American pika including climate change, livestock grazing, invasive plant species and fire suppression. Largely due to sensitivity to temperature (death can occur after brief exposures to ambient temperatures greater than 77.9 °F), climate change was determined to be the primary threat to the species (FWS 2010).

Pikas appear to have a narrow niche in regards to suitable habitat and thermal tolerance. Although the rocky high elevation habitats occupied by pikas are generally not at risk of loss or substantial alteration, changes to the snow pack and ambient temperatures could decrease the suitability of these areas. Considering this narrow niche, three of the most pressing threats to the persistence of pikas include changes in temperature, reduction in snow pack, and challenges to dispersal. Beever et al. (2010) suggest that chronic heat stress and acute cold stress may play an important role in pika extirpation and/or movement to new areas. Chronic heat stress is associated with an increased number of days during the summer with higher temperatures – which can negatively impact pikas directly or reduce the amount of time available for foraging (Beever et al. 2010, Holtcamp 2010). Acute cold stress can kill animals directly and may be increasing as the snowpack is reduced or lost. Without the insulating effects of the snow overhead, pikas can be exposed to lethal freezing temperatures in their rock burrows during the winter (Beever et al. 2010). Evidence of declines in lower elevation populations – potentially due to climate change – has been observed within the Great Basin where 7 of 25 populations (28%) reported earlier in the 20th century appeared to have experienced recent extirpations (Beever et al. 2003).

Despite their limited thermal tolerance and close association with alpine habitat, there is disagreement about the degree to which climate change poses a threat to the American Pika as a species. In a recent rapid-assessment of potential pika sites conducted by Millar and Westfall (2010), the authors found that pika occur in lower and warmer sites than previously documented. Similar to Beever et al. 2003, Millar and Westerfall (2010) found evidence of local extirpations in central Great Basin sites with 50% of sites being categorized as “old” (previously - but not currently – used by pika). However, there was no association between the age of sites and elevation, indicating that lower and warmer sites are not being extirpated at a faster rate than higher elevation sites, thus climate change may not be the primary driver of declines. Of the pika sites assessed within the Sierra Nevada range only 2% of sites were categorized as “old”, suggesting that the Sierra Nevada pika populations are not experiencing the same declines as those within the Great Basin. Additionally, a Fish and Wildlife assessment of the American pika concluded that although the species could potentially be impacted by climate change, there remains enough high elevation habitat to ensure the species’ long-term survival (FWS 2010).

Finally, although pikas generally specialize in remote habitats at high elevations in mountainous terrain, their dispersal abilities are quite poor. As habitat becomes unsuitable with changes in temperature and snowpack, pikas will have difficulty colonizing new areas as compared to more mobile species. Thus any obstacles that would



decrease pika's already limited dispersal ability or reduce connectivity of suitable habitat could be considered a threat. Some high elevation sites are naturally fragmented, but in other cases the addition of roads, development, predators, etc. may further impede a pika's ability to disperse successfully to more hospitable areas, however these conditions do not occur within SEKI (i.e., anthropogenic habitat fragmentation is not a concern).

### ***Conservation in Sequoia and Kings Canyon***

Sequoia and Kings Canyon National Parks encompass large areas of undisturbed alpine and subalpine habitat in the eastern halves of the parks (Figure A1.3), including talus slopes and boulder fields that can provide suitable habitat for pikas (Sequoia-Kings Canyon NPS observation database). Sequoia and Kings Canyon National Parks may play a unique role in the conservation of pikas in the southern Sierra Nevada because they contain some of the highest elevation mountains in the lower forty-eight states, include several of the highest peaks in the Sierra Nevada, and harbor some of the southernmost populations of pikas in North America. In particular, the Mount Whitney subspecies appears confined to the southern Sierra Nevada and is the primary subspecies found within the parks. Pikas that occur within the parks may have opportunities to shift upward in elevation or move northward where mountains are connected by suitable habitat.

No particular actions have yet been taken to maintain pika populations within the parks – although park biologists have encouraged park employees to document occurrences of pikas for submission to the observation database (e.g., backcountry rangers have made numerous contributions of pika sightings recently). However, due to a lack of non-climatic anthropogenic threats to the American pika, little direct management has been needed up to this point. If climate change proves a major threat to pika within Sequoia and Kings Canyon, more direct management actions may be desired. However, tools for helping pika adapt to climate change remain limited. Possible exceptions include aiding dispersal to cooler and higher habitat through construction of rock walls between fragmented talus slopes (per Millar 2010) or translocation.

### ***Data Gaps & Research Priorities***

Recent surveys suggest that pika is distributed more widely than previously thought in the Sierra Nevada (Millar and Westfall 2010). This evidence challenges current knowledge of the species' range and abundance (i.e., Table A1.2) and suggests that more complete assessments are needed to fully understand the distribution and condition of pika populations. Additionally, historical records of species presence and absence (e.g., Grinnell 1924) demonstrate that periodic surveys can be invaluable for assessing a species' response to a changing climate.

Scientists' ability to accurately forecast local temperature and precipitation changes due to climate change remains limited. This is especially true in areas of high topographical variability such as the Sequoia and Kings Canyon National Parks. Furthermore, such coarse projections overlook microclimates caused by geographic variation within the landscape, which have the potential to act as climate refugia for species such as pika. For example, rock-ice feature till found within rock matrix environments allow for cooler than expected summer temperatures and warmer than expected winter temperatures. Such conditions may provide favorable habitat for pika at lower elevations than otherwise

expected (Millar and Westfall 2010). Further research into how such refugia function and where they may be located within Sequoia and Kings Canyon National Parks would help researchers and managers predict where pika are most likely to persist despite general regional warming.

### ***Summary and Recommendations***

The American pika is distributed throughout many of the mountain ranges of western North America. Within the species are 36 recognized subspecies, two of which (Yosemite pika and Mt. Whitney pika) are found within the Sequoia and Kings Canyon National Parks. These subspecies are currently considered vulnerable by the NatureServe conservation ranking system. The influence of a changing climate (temperature, precipitation) in combination with the physiological constraints of pika will likely be the most influential driver contributing to its vulnerability and to future shifts in distribution and abundance of pika within the parks.

Within the Sequoia and Kings Canyon National Parks recorded observations indicate pikas are well distributed across high elevations. However, a systematic assessment would provide a more complete picture of where this species occurs. Furthermore, an on-going monitoring program would allow for a better understanding of how pikas respond to climate change. If systematic monitoring reveals a contraction in the distribution of pika at the lower extent of their elevation range, future management options are somewhat limited, but could include increasing connectivity between fragmented habitat and translocation of individuals to cooler sites.

### ***Data Sources***

Species life history information was taken from a combination of NatureServe species accounts, California department of Fish and Game reports, United States Fish and Wildlife Service reports and the primary literature. Distribution data was acquired for North America, California and Sequoia and Kings Canyon National Parks from NatureServe, CNDDDB and the SEKI observation database respectively. An evaluation of the current condition of the American Pika and threats to the species was compiled using the US Department of Fish and Wildlife reports, and the primary literature (e.g., Beaver et al. 2003 and Millar and Westfall 2010).

## A.2. Sierra Nevada Red Fox – *Vulpes vulpes necator*

### Abstract

The Sierra Nevada red fox (*Vulpes vulpes necator*, SN red fox) a rare subspecies of the widespread red fox (*Vulpes vulpes*). The SN red fox occurred historically in the higher elevations of Sequoia and Kings Canyon National Parks. Little information is available on the ecology of this subspecies, but limited historic and recent accounts suggest it is primarily associated with subalpine forest, high elevation meadows, and alpine rocky habitats, with some use of middle elevation forests in the winter. Verifiable records for the region include individuals trapped east and southeast of Sequoia National Park in the vicinity of Mount Whitney and Whitney Meadows. The Parks Wildlife Observation database includes 14 sightings between 1934 and 2005. Three sightings are from the 1950s, four from the 1960s, one from the 1990s and three from the 2000s. The current status of this subspecies in the parks is uncertain, as it has gone undetected in recent surveys. However, recent verification of several SN red foxes north of Yosemite National Park suggests this subspecies is capable of persisting undetected for long periods of time.



**Figure A2.1.** Recent photo of a cross-phase Sierra Nevada red fox taken at a remote camera station in 2010 near Sonora Pass, California (courtesy of S. Lisius, US Forest Service).

- **Species global status:** G5T1T3 (globally secure with imperiled local populations), Rounded global status for sub-species is T2 (imperiled).
- **Park status:** Rare to non-extant. Historic records are predominantly from high elevation barren and subalpine habitats. There have been no verifiable records (e.g., photo, specimen, DNA) in the parks for well over 50 years.
- **Overall integrity:** Very low. The current status of this species in the parks is unknown, but even historic accounts suggest that densities were generally low. Habitat integrity may be moderate to high – as the apparently preferred subalpine forest, alpine meadow, and barren alpine habitats are mostly undisturbed with the exception of grazing and recreation activities around high elevation meadows.
- **Certainty of integrity:** Low. Historic information is limited and recent surveys have not detected this subspecies in or around the parks. However, as the Sierra Nevada red fox lives in remote areas and was historically described as being wary of humans, it may have simply avoided detection – as was the case with the remnant population detected in 2010 near Sonora Pass (US Forest Service 2010).
- **Metric to evaluate integrity:** Results of surveys (Zielinski et al. 2005, Green 2007, Hudgens and Garcelon 2007) and historic accounts (Grinnell et al. 1937).
- **Vulnerability with respect to key stressors:** If the Sierra Nevada red fox still occurs in the Parks, climate change could be a stressor due the subspecies' close association with subalpine and alpine zones. There is also concern that grazing of sheep and pack animals in alpine meadows has altered vegetation and prey base.

### **Background and Taxonomy**

The red fox (*Vulpes vulpes*) is the most geographically widespread carnivore in the world (Larivière and Pasitschniak-Arts 1996). This medium-bodied canid is distributed across much of Europe, Asia, North America, and northern Africa, but has also been introduced to areas where it did not occur previously (Grinnell et al. 1937; Larivière and Pasitschniak-Arts 1996). Although many subspecies of red fox appear to be maintaining or even expanding their distributions, the range of the Sierra Nevada red fox (*Vulpes vulpes necator*) has been drastically reduced over the last century. Trapping, poisoning, and probable habitat loss have contributed to the decline of the Sierra Nevada red fox and it is now considered one of the rarest mammals in California (Larivière and Pasitschniak-Arts 1996). Historically, the Sierra Nevada red fox – often referred to as the mountain red fox – was known to occur at low densities in middle to high elevations of the Sierra Nevada. Until recently, the only population of this subspecies known to still exist in California was located in the vicinity of Lassen National Park in the northeastern portion of the state (Perrine et al. 2010). However, in August of 2010, photographs of what appeared to be a red fox were taken at a baited camera station near Sonora Pass north of Yosemite National Park; genetic and additional photo evidence have now verified the presence of at least two Sierra Nevada red foxes in this portion of the historic range of the subspecies (S. Lisius, pers. com., US Forest Service 2010, Figure A2.1). As of this writing, several additional confirmed sightings of Sierra Nevada red foxes have also occurred in the region, although none of them have been within SEKI.

### **Physical Description**

The Sierra Nevada red fox is a relatively small and slender bodied canid with a long muzzle, large pointed ears, and a bushy tail that is nearly as long as its body (Figure A2.1,A2.2; Grinnell et al. 1937; Larivière and Pasitschniak-Arts 1996). Pelage coloration in red foxes can be variable with two notable exceptions: the tail always has a white tip and the backs of the ears are always black. Red foxes have three color phases – red, silver/black, and cross – all of which have been documented in the Sierra Nevada. Red phase foxes are the most common; these

individuals have red to reddish-blond fur on the head, back, and sides, white fur on the throat and belly, and black fur on the legs (Figure A2.2). The fur of silver or black phase individuals varies from a silver-gray to almost black. Cross phase individuals are grayish-brown overall, with a concentration of black guard hairs forming a “cross” over the back and shoulders (Figure A2.1). In California, the red fox overlaps in distribution with coyote and gray fox, which can contribute to misidentification of these species. Coyotes are larger and more blondish-brown on the head, back, and legs, have rust-colored fur behind the ears and a black tipped tail that is half the total body length. Gray foxes are generally smaller, have salt-and-pepper colored fur on their head and backs,



Figure A2.2. Radio-collared Sierra Nevada red fox

[www.nps.gov/lavo/naturescience/mammals](http://www.nps.gov/lavo/naturescience/mammals)

rusty red fur around the neck, backs of ears, sides, and legs, and white fur on the throat and belly. Gray foxes have a bushy tail that is nearly as long as their body length, but the black tip distinguishes this species from red fox. Red foxes across North America range in weight from 3.5 to 7 kg (Ables 1975; Voigt 1987); weights of Sierra Nevada red foxes from the Lassen Peak area were 3.5 kg for females ( $n = 4$ ) and 4.0 kg for males ( $n = 1$ ; Perrine 2005).

### **Taxonomy and Genetics**

The Sierra Nevada red fox is in the Kingdom *Animalia*, Phylum *Craniata*, Class *Mammalia*, Order *Carnivora*, Family *Canidae*, and Genus *Vulpes* (NatureServe 2010; Hall 1981). There are over forty subspecies of red fox (*Vulpes vulpes*) globally, of which the Sierra Nevada red fox (*V. v. necator*) may be the rarest. In California, there has been some debate as to which red fox populations are native and which were introduced by humans. After extensive surveys, interviews with trappers, and examination of collected specimens, Grinnell et al. (1937) suggested that the red foxes which live chiefly above 2,130 m (7,000 ft) in the Sierra Nevada were a unique subspecies based on their distinct habitat preferences, distribution, and morphology. This subspecies distinction, referred to as the “mountain” or Sierra Nevada red fox, has since been supported with genetic evidence (Perrine et al. 2007). Perrine et al. (2007) also compared genetic samples from “mountain” red fox museum specimens with recent samples collected near Mount Lassen and found compelling evidence of the persistence of the Sierra Nevada red fox subspecies in this geographic area. Genetic analysis of DNA samples from the recent detections near Sonora Pass by Dr. Ben Sacks of UC Davis further support the existence of remnant populations of this rare subspecies in California (US Forest Service 2010).

Grinnell et al. (1937) discussed a population of red foxes living in the lowlands of the upper Sacramento Valley in the late 1800’s and early 1900s. Due to its location, the authors believed the Sacramento Valley population was introduced by humans, although they could find no records to confirm this. Interestingly, recent genetic work suggests that red foxes of the upper Sacramento Valley are more closely related to the Sierra Nevada subspecies than to other red fox populations in California (Perrine et al. 2007; Sacks et al. 2010). This suggests that both the “mountain” and upper Sacramento Valley populations are native to the state – but gene flow has been restricted or nonexistent in recent history. Other lowland red fox populations in California (e.g., San Francisco Bay, southern California) have likely resulted from fur farm escapees and deliberate introductions; these populations have expanded substantially over the last fifty years and have begun to cause problems for a number of rare species including nesting seabirds and San Joaquin kit fox (Lewis et al. 1999).

### **Life History**

*Habitat.* Across its global distribution, the red fox (*Vulpes vulpes*) occurs in a wide variety of habitats, is generally adaptable to fragmented landscapes, and has survived on lands in close proximity to humans despite the pressures of hunting, trapping, and other forms of persecution. The Sierra Nevada subspecies is somewhat uncharacteristic of red foxes in general in that it occupies a relatively narrow habitat niche, occurs in mountainous environments, and has persisted only in remote areas. Historic records of the Sierra Nevada red fox are largely from above 2,130 m (7,000 feet) in the Canadian



and Hudsonian life zones of the Sierra Nevada (Grinnell et al. 1937). Habitat types associated with old sightings, tracks, and trapping records include mixed conifer forest at middle elevations and subalpine forest, talus, meadows, and exposed barren rock at high elevations (Grinnell et al. 1937). Trappers and naturalists also reported that Sierra Nevada red foxes appeared to move downslope for the winter – perhaps in search of more accessible food resources and/or to avoid deep snow (Grinnell et al. 1937).



**Figure A2.3.** Photo of a Sierra Nevada red fox at a baited camera station near Sonora Pass, California in 2010 (courtesy of S. Lisius, US Forest Service).

A recent study near Mount Lassen detected red foxes at remote camera stations in barren, conifer, and shrub dominated habitats at relatively high elevations (~1,400 to 2,600 m range; Figure A2.3; Perrine 2005, Perrine et al. 2010). An associated study involving radio collared animals (1 male, 3 females) in the same geographic area revealed a general pattern of foxes selecting barren (exposed rock) habitats, avoiding mid-elevation conifer and shrub habitats, while using high elevation conifer forest in proportion to availability (Perrine et al. 2010). In winter, radio-collared foxes were documented using Sierran mixed conifer, red fir, montane chaparral, and white fir habitat types. At the

rest site scale, radio-collared red foxes were found using clusters of small red firs, spaces amongst boulders in talus slopes, openings in shrub patches, subnival cavities under logs and trees, and sheltered spaces formed by snow covered trees (Perrine 2005; Perrine et al. 2010). Benson et al. (2005) tracked radio-collared red foxes in the snow near Mount Lassen and found that they often traveled in the forest around edges of meadows as opposed to crossing wide openings and occasionally traveled on ski and snowshoe tracks – apparently taking advantage of the compacted snow for ease of travel. Although data was limited, Benson et al. (2005) found that red foxes used forest cover more than expected compared to open areas during the winter.

Historical records and recent studies in the Lassen area are of great value in that they provide at least some information on the habitat associations of the Sierra Nevada red fox. However, it is important to note that the paucity of records and studies from other parts of the Sierra Nevada, particularly in recent years, prevent extensive interpretation of the habitat needs of the subspecies across its former range. It is unclear whether lands used by the remaining red foxes near Mount Lassen represent optimal habitat or simply areas where the species has been able to persist due to other factors (e.g., inaccessibility to trappers). On the other hand, there is much agreement between the habitats noted in historic records and those used by red foxes in recent studies near Lassen, suggesting that high elevation forests and barren areas likely are important in summer while middle to high elevation forests may be used more frequently in winter.

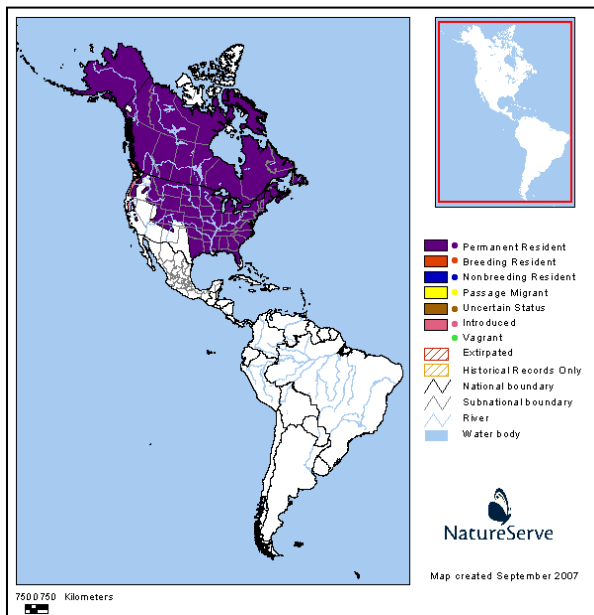
**Diet.** Globally, red foxes are known as adaptive generalists and have a varied diet which includes lagomorphs, sciurids, small fossorial mammals, fawns, galliforms, ducks, insects, reptiles, garbage, and occasionally other small carnivores (Larivière and Pasitschniak-Arts 1996). Historic records mention foods either known or suspected to be eaten by the Sierra Nevada red fox including: white-tailed jackrabbit (*Lepus townsendii*), bushy-tailed woodrat (*Neotoma cinerea*), Douglas squirrel (*Tamiasciurus douglasii*), Belding ground squirrel (*Spermophilus beldingi*), pika (*Ochotona princeps*), golden ground squirrels (*Spermophilus lateralis*), voles (*Microtus* sp.), blue grouse (now the sooty grouse, *Dendragapus fuliginosus*), and various species of chipmunk (*Tamias* sp.), mice, and songbird (Grinnell et al. 1937). A recent study near mount Lassen reported that red fox scats contained pocket gophers (*Thomomys monticola*), mice (*Peromyscus* sp.), voles (*Microtus* sp.), and ground squirrels (*Spermophilus* sp.) most frequently, but also found evidence of carrion, insects, manzanita berries, birds, and garbage on a seasonal basis. Interestingly, lagomorphs were absent from scat samples, suggesting they may no longer be abundant and/or available in this geographic area.

**Reproduction and Survival.** There is little specific data available on reproductive ecology or survival rates of the Sierra Nevada red fox subspecies. Red foxes typically breed in winter, gestation lasts almost 2 months, and pups are born in spring (NatureServe 2010). Litter size ranges from 4 to 5 and adult males likely assist in raising the young – a behavioral pattern common within other subspecies (NatureServe 2010). Dens are suspected to be located in rock slides at high elevations, but further research is needed (NatureServe 2010).

### Distribution

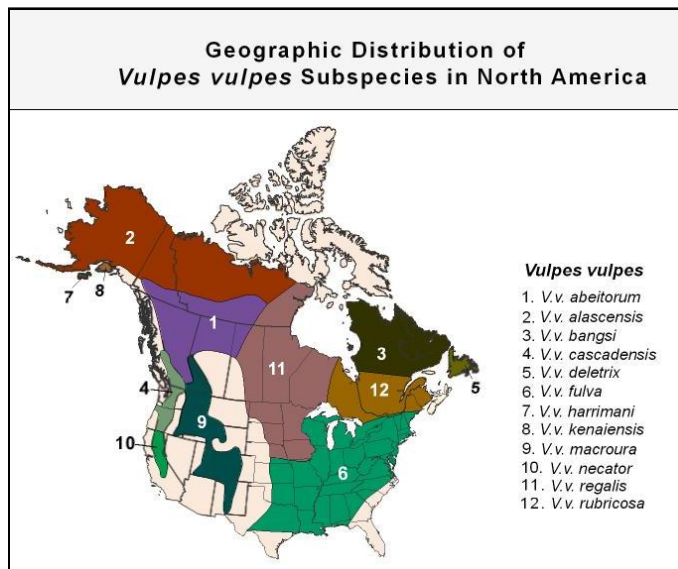
The red fox (*Vulpes vulpes*) is broadly distributed around the globe – predominantly in the northern hemisphere in Europe, Asia, North America (Figure A2.4), and northern Africa. The distribution of the Sierra Nevada subspecies (*Vulpes vulpes necator*) is poorly known, but historically included the Sierra Nevada of eastern California and a portion of western Nevada (Figure A2.5). The closest neighboring subspecies is the Cascades red fox which occurs further north in the Cascade Mountains of Oregon and Washington (*Vulpes vulpes cascadenis*). In California, historical records of the Sierra Nevada red fox come from the Mount Lassen area in the northeastern part of the state south to the vicinity of Mount

Whitney in Sequoia National Park in the southern Sierra Nevada (Grinnell et al. 1937, Schempf and White 1977). Records are predominantly from higher elevations (~2,100 m



**Figure A2.4.** Distribution of the red fox (*Vulpes vulpes*) in North America (NatureServe 2007).

and up) of the Sierra Nevada and red foxes reportedly occurred on both the west and east sides of the range.



**Figure A2.5.** Distribution of the twelve subspecies of red fox (*Vulpes vulpes*) in North America as presented by Ables (1975).

Until quite recently, the only native mountain red fox population known to still occur in California could be found in and around Lassen National Park (Perrine 2005). The Lassen population appears to be small and isolated, but does inhabit a geographic area historically occupied by the Sierra Nevada subspecies. In August of 2010, an individual red fox was detected by US Forest Service Biologists (Sherri Lisius, Adam Rich) at a baited camera station near Sonora Pass on the Humboldt-Toiyabe National Forest in the northern Sierra Nevada (US Forest Service

2010). Dr. Ben Sacks of UC Davis was able to confirm the individual as a Sierra Nevada red fox using DNA from saliva and hair collected at the camera site. Since the initial detection, several individuals have been identified in this area; although this rediscovery happened quite recently and implications for the local population are still unclear, it suggests that remnant native red fox populations may persist in other parts of the historic range outside of the Lassen area.

Grinnell et al. (1937) reported a number of red fox specimens, records, and observations from the vicinity of Sequoia-Kings Canyon and Yosemite National Parks (Figure A2.6). The original type specimen for the subspecies was collected by Merriam (1900) at Whitney Meadows, just east of Sequoia National Park (9,500 ft). Two other specimens (females) were collected in 1911 in the same area (9,800 ft). Other specimens were collected at Monache Meadows (southeast of Sequoia NP) and at Saddlebag, Ellery, and Virginia Lakes (west and northwest of Yosemite NP). Other records reported from within Sequoia National Park include: Wallace Creek, Lake South America, and the headwaters of the Kern River – all high elevation sites (~3,000 – 3,400 m) in the northeastern portion of Sequoia National Park. Of note, recent surveys conducted within the Parks had survey sites in the vicinity of all three of these locations, but no red foxes were detected (Green 2007).

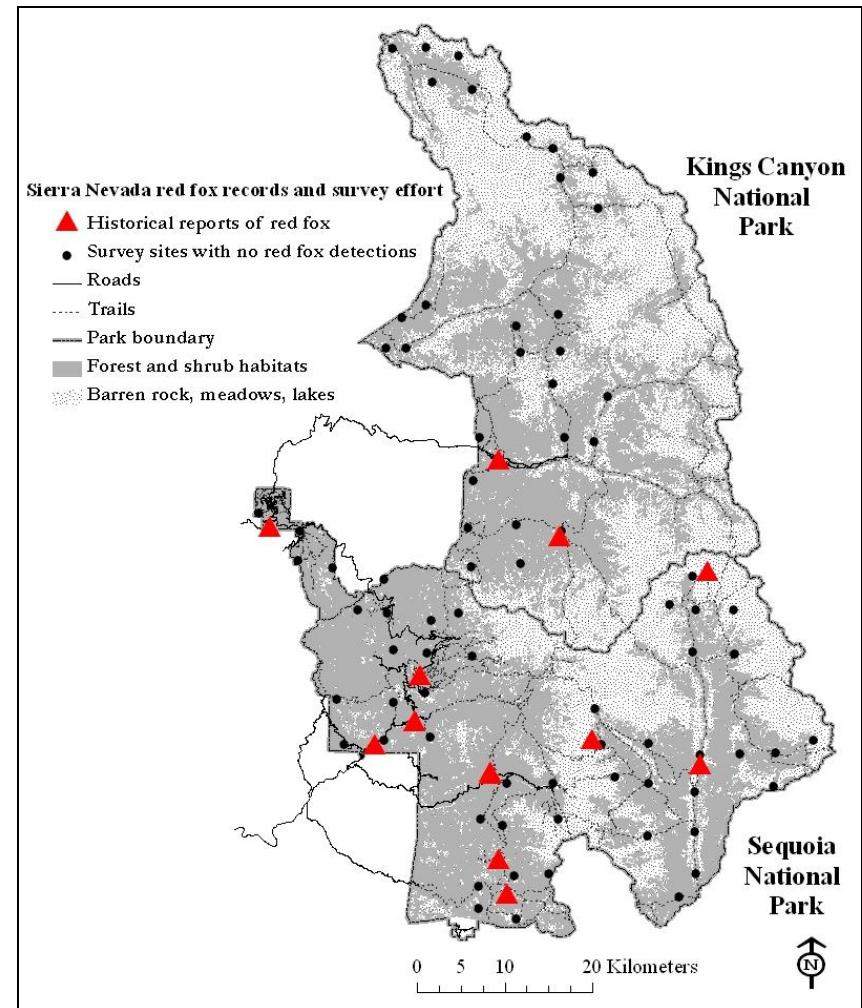
Records from other NPS Park reports and the wildlife observation database are relatively sparse, but date all the way back to the 1930's (Figure A2.7). These records include sightings and/or tracks in the vicinities of the following areas: Lake South America, Little Five Lakes, Rock Creek, Mineral King, Hockett Meadow, Scaffold Meadows, South Fork



Meadows, General's Highway, Grant Grove, and Cedar Grove. As gray foxes can easily be mistaken at a quick glance for a red fox due to the red fur around their face and sides and the tracks of gray fox, coyote, and red fox could be difficult to differentiate, it is hard to know whether the observational records are entirely accurate. A more concrete record comes from November of 1940 when Ranger Shellenberger caught a cross or black morph red fox just south of Sequoia NP in lower Rocky Basin (3,000 m). The same ranger reported likely red fox tracks within the Parks and noted that red foxes were likely to occur in the vicinity of the Siberian Outpost.



**Figure A2.6.** Historical distribution of the Sierra Nevada red fox (*Vulpes vulpes necator*) in California (from Grinnell et al. 1937). The orange star indicates the general locality of Sequoia-Kings Canyon National Parks. The blue star shows the general area of recent detections near Sonora Pass.



**Figure A2.7.** Distribution of unconfirmed reports of Sierra Nevada red fox from the parks' observation database and location of sites surveyed for carnivores (2002-2004) with no red fox detections in Sequoia-Kings Canyon National Parks. As gray foxes can be mistaken for red foxes, records from lower elevations in the west half of the parks may be less reliable than those in subalpine and alpine habitats of the middle and east half of the parks.

### **Status**

The Sierra Nevada red fox (*Vulpes vulpes necator*) was listed as a threatened subspecies under the California Endangered Species Act by the Department of Fish and Game in 1980 (Gould 1980). After a five year status review (California Department of Fish and Game 1987), the listing was still deemed as warranted and this subspecies is currently listed as threatened under California guidelines. The global conservation status of the Sierra Nevada subspecies is considered vulnerable due to the uncertainty associated with its current distribution and potential vulnerability to timber harvest, grazing in alpine meadows, use of rodenticides, and recreation (NatureServe 2010, Perrine et al. 2010).

### **Threats**

Historically, the major threats to the persistence of the Sierra Nevada red fox were fur trapping, poisoning, and grazing in alpine meadows. When Grinnell et al. (1937) summarized information on furbearers in California, mountain red foxes were already rare; although this subspecies was caught infrequently, fur-trapping likely had a significant impact on the size and distribution of populations that have persisted in the state (Grinnell et al. 1937). The practice of leaving poison in sheep carcasses is also suspected to have had a substantial impact on red fox and other carnivore populations living at middle to high elevations. This custom was developed by sheep herders in an effort to reduce predation on their livestock and likely resulted in the deaths of untold (and unrecorded) numbers of carnivores in the Sierra Nevada (Grinnell et al. 1937). Grinnell and his coauthors (1937) suspected that sheep grazing in high elevation meadows may have also impacted red foxes indirectly by reducing the forage available for their prey, thus reducing prey abundance. As trapping for Sierra Nevada red fox is now illegal and rodenticide is likely the only poison being used within the range of this subspecies, other threats appear to have become more pressing in modern times.

Currently, a major concern related to the Sierra Nevada red fox in California is the potential for hybridization with introduced subspecies. Based on recent work, the Lassen population of mountain red foxes appears to still be genetically unique, but may have reduced genetic diversity (Perrine et al. 2007). Small population size is another major concern – as it increases the potential for the subspecies to succumb to disease, natural or human-based catastrophic events (e.g., fire), or other alterations to the landscape (e.g., roads, fragmentation). Limited genetic diversity also has the potential to decrease resistance to disease or cause additional health problems (Mills 2007). As with many species that occur predominantly at higher elevations, global climate change could have a negative impact on prey availability, distribution of suitable habitat, and environmental conditions that favor the mountain red fox (Perrine et al. 2010). Finally, there is so little information available on habitat use and suitability that it is difficult to accurately assess habitat related threats that may have developed over time.

Much of the habitat in the vicinity of historical records in the Parks remains relatively pristine compared to other parts of the Sierra Nevada. However, timber harvest, grazing in alpine meadows, habitat fragmentation, human activity, fires, and recreation in or around the Parks could potentially have negative impacts on mountain red foxes in this geographic area.

### ***Data Gaps, Research Priorities, and Conservation***

***Critical Data Gaps.*** Very little is known about the Sierra Nevada red fox, including whether or not it still occurs in the Parks. Over the last ten years, at least three different projects focused on assessing carnivore distributions have been conducted in the vicinity of Sequoia and Kings Canyon National Parks using devices that could have detected this elusive carnivore (Zielinski et al. 2005, Green 2007, Hudgens and Garcelon 2007). It is worthwhile to note that the older model remote cameras used in these studies lacked many of the improved features of digital cameras available today; whether this contributed to lack of detection is unknown, but certainly not inconceivable. As with wolverine (*Gulo gulo*), it is possible that a small population of red foxes still exists in a remote part of the southern Sierra Nevada, however, the continued lack of verifiable detections is not a promising sign for the existence of a viable population in the region. If the Sierra Nevada red fox has managed to survive undetected in the Parks or if it could be reintroduced, some of the basic data gaps that could assist conservation efforts include: habitat requirements, home range size, seasonal movement patterns, demographic parameters, level of genetic diversity, susceptibility to disease or disturbance, details of reproductive ecology, and an understanding of any specific vulnerabilities associated with this geographic area. Some generalizations can be drawn from historic records and studies in other geographic areas, but the lack of extensive information on the ecology and former distribution of this subspecies in the Parks combined with its tendency to inhabit remote areas certainly make active conservation of this elusive animal challenging.

***Climate Change.*** Based on what is known about this subspecies and the knowledge that it typically occurs at higher elevations, it seems likely that climatic change could negatively affect the Sierra Nevada red fox (Perrine et al. 2010). Warmer temperatures in the Sierra Nevada are expected to shift vegetative communities upward in elevation (Lenihan et al. 2003, Hayhoe et al. 2004), which could not only reduce the total area of suitable habitat, but also decrease availability of potential prey in alpine areas (e.g., white-tailed jackrabbit, pika) and alter winter snow conditions in a manner that might not be advantageous for SN red fox.

### ***Summary and Recommendations***

Because it is unknown whether or not the Sierra Nevada red fox still occurs in the southern Sierra Nevada, it is difficult to make recommendations that would promote the long-term persistence of this species in Sequoia-Kings Canyon National Parks. Although historical threats have become less relevant, they have had a lasting impact on the distribution of red foxes in the Sierra Nevada. Current potential threats, such as small population size, reduced genetic diversity, disease, and global climate change, may be difficult to address – particularly with an elusive animal that dwells in remote areas. Also, as the habitat types and elevation ranges that were historically suitable for Sierra Nevada red fox in the Parks have been largely protected for over a century, there are not obvious management changes that could improve current conditions within Park boundaries except perhaps to limit grazing and/or recreational activities in a specific area if a red fox is detected. A reintroduction effort could be effective in the Parks, but currently there are no known sustainable populations to use as a source. Thus, active conservation of the Sierra Nevada red fox in the Parks may not be an achievable goal at

this time. A long-term monitoring program designed to detect multiple species, including red fox, might be the best option for now; and with the recent detections near Sonora Pass (US Forest Service 2010), this approach certainly warrants further consideration.

Several small to medium-bodied carnivores of concern occur or have occurred historically within the Parks (e.g., wolverine, fisher, American marten), so periodic monitoring for this suite of animals may be justifiable for a number of reasons. Survey devices are available that are capable of detecting multiple species, such as track plates, remote cameras, and hair snares. If systematic carnivore surveys are not economically or logistically feasible, the operation of remote camera stations by backcountry rangers during the summer, snow survey personnel during the winter, and/or outside researchers anytime during the year could provide a means of periodically searching for this subspecies with a relatively small amount of effort. High quality remote digital cameras with long lasting battery capabilities are more affordable than ever, have improved resolution, and could provide verifiable evidence of a red fox – especially if used in conjunction with a hair snare or other device to collect DNA material. If a red fox were to be detected, a more extensive research and/or monitoring plan specifically targeting this subspecies in a defined portion of the parks could then be implemented.

#### ***Data Sources***

Information on the life history and distribution of this rare subspecies was compiled from NatureServe Explorer species accounts, the dissertation and related publications of John Perrine, other available publications associated with red fox in California, historic records, the Parks' wildlife observation database, and reports from the US Forest Service and California Department of Fish and Game. Although relatively old and somewhat limited by data sources, the historic records and natural history described in Grinnell et al. (1937) still provide some of the best information available on this subspecies in the vicinity of Sequoia and Kings Canyon National Parks and the Sierra Nevada in general. The recent verification of several individual Sierra Nevada red foxes near Sonora Pass on the Humboldt-Toiyabe and Stanislaus National Forests will certainly spur investigation and research which may prove useful to Sequoia-Kings Canyon. Future useful contacts for updates on the subspecies in this area include:

- Sherri Lisius, Wildlife Biologist, Humboldt-Toiyabe NF, Bridgeport office
- Adam Rich, Wildlife Biologist, Stanislaus NF
- Diane Macfarlane, Wildlife Biologist, Pacific Southwest Region Threatened, Endangered and Sensitive Species Program, US Forest Service, Vallejo office
- Dr. Ben Sacks, Professor, Canid Diversity and Conservation Unit, UC Davis
- Dr. John Perrine, Professor, Biological Science Dept., CalPoly State University

### A3. Wolverine – *Gulo gulo*

#### **Abstract**

The wolverine (*Gulo gulo*) is a large carnivorous mammal that is uncommon due to low natural densities and high sensitivity to human disturbance. Wolverines were considered to be extirpated from California by 1922 and the presence of a viable population in the southern Sierra Nevada range is highly unlikely (Garcelon et al. 2009). Due to the availability of large tracts of protected habitat within the park and in adjacent federal lands, translocation of wolverines from other populations is possible. This species requires large tracts of undisturbed habitat, so habitat degradation outside of SEKI could affect species persistence within the parks, and a formal attempt to re-establish wolverine would require interagency cooperation, and may require repeated attempts.

- **Species global status:** Secure (NatureServe), Vulnerable (IUCN)
- **Park status:** Likely extirpated or, if present, at very low densities
- **Overall integrity:** Low
- **Certainty of integrity:** High
- **Metrics to evaluate integrity:** Recent surveys for wolverines and other carnivores in SEKI and nearby areas have not found any evidence of wolverine presence.
- **Vulnerability with respect to key stressors:** Within the park, individuals (if present) may be vulnerable to disturbance from recreation. Resource extraction and human development outside of the parks may pose threats to long-term persistence in the region. Climate change is likely to negatively impact this species due to projected declines in spring snow cover.

#### **Description and taxonomy**

The wolverine (*Gulo gulo*, family Mustelidae) is a large carnivorous mammal found in boreal and mountainous regions in North America and Eurasia (Figure A3.1). Despite its wide geographic distribution, the wolverine is uncommon because of naturally low densities and sensitivity to human disturbance (Copeland and Whitman 2003). Wolverines have a bear-like stature, with thick, muscular limbs and large, hairy feet that allow them to walk easily in deep snow. The largest terrestrial mustelids, wolverines can range in size from 8-18 kg, with males typically weighing 40-60% more than females (Banci 1994, Copeland and Whitman 2003). Wolverines are known for being both ferocious and strong.



**Figure A3.1** Photo of wolverine  
([http://en.wikipedia.org/wiki/File:Wolverine\\_on\\_rock.jpg](http://en.wikipedia.org/wiki/File:Wolverine_on_rock.jpg))

The wolverine is in the kingdom *Animalia*, phylum *Craniata*, class *Mammalia*, order *Carnivora*, family *Mustelidae*, and genus *Gulo* (NatureServe 2009). Currently, most taxonomists are in agreement that Eurasian and North American wolverines are the same

species, *Gulo gulo*. Copeland and Whitman (2003) listed two recognized subspecies, *G. g. gulo* found in Eurasia and *G. g. luscus*, found in North America. California wolverines were genetically distinct from wolverines known from other areas of their range, likely due to the geographic isolation of alpine habitat in the Sierra (Schwartz et al. 2007).

### **Life History**

Wolverines tend to occupy mountainous and alpine habitats. In the southern Sierra, wolverines are known to have used a range of habitats from 2000-3400 m elevation that include alpine shrublands, mid to high elevation conifer forest types and montane chaparral (Johnson 1988-1990). Wolverine are highly sensitive to human disturbance and long-term viability of populations may be dependent on the availability of large tracts of undisturbed habitat.

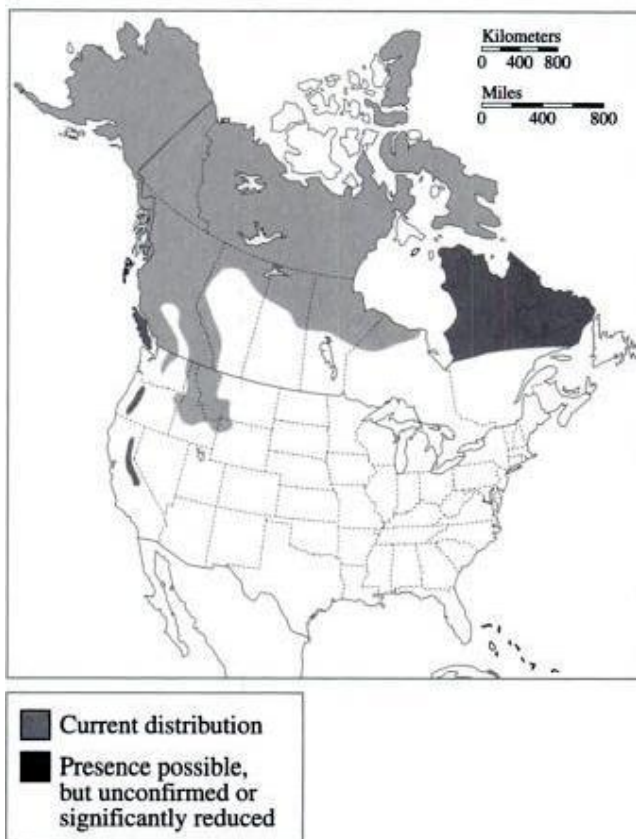
Wolverines are opportunistic feeders. Diet studies of wolverines in Alaska and Canada indicate that wolverines largely consume scavenged ungulate carcasses, such as moose (*Alces alces*) and caribou (*Rangifer tarandus-caribou*), but may also hunt small and medium sized prey items, such as small rodents, ground squirrels, porcupines (*Erethizon dorsatum*), red fox (*Vulpes vulpes*), hares (*Lepus* spp.), sheep (*Ovis* spp.) and ptarmigan (*Lagopus* spp.) (Copeland and Whitman 2003, NatureServe 2009). No diet studies have been done specifically on wolverines in California, but the diets of wolverines in the Sierras probably relied heavily on bighorn sheep (*Ovis Canadensis*) and mule deer (*Odocoileus hemionus*) (Garcelon et al. 2009).

Wolverines do not hibernate, have large territories (48-2000 km<sup>2</sup>) and are known to move large distances in search of food, new territories, and mates (Banci 1994, Vangen et al. 2001). They are solitary and tend to exclude others of the same sex from their territories. Fitting with their polygamous mating system, males typically have very large territories that overlap with multiple smaller female territories (Banci 1994, Copeland and Whitman 2003). Adults become sexually mature at age 2-3 (Rausch and Pearson 1972). They breed from May-June and typically have litters of 2-4 kits, born in late winter or early spring (NatureServe 2009). Kits are raised in snow dens that may be associated with trees or boulders, depending on the habitat characteristics (Magoun and Copeland 1998). Snow cover for denning habitat is critical for reproduction and may be a limiting factor in their distribution (Aubrey 2007).

### **Distribution**

*Global.* Wolverine are globally widespread across boreal regions, with isolated populations toward the south in montane habitats (Figure A3.2). Global population size is unknown, but is likely to be in the hundreds of thousands (NatureServe 2009). Global populations are stable to declining, depending on the geographic region. In North America, the largest populations are found in Canada and Alaska (Banci 1994, Copeland and Whitman 2003, NatureServe 2009). The Yukon populations are relatively stable due to minimal disturbance from humans. In contrast, the wolverine populations in the contiguous US have been declining since the 1800s, largely due to trapping and habitat disturbance by humans (Copeland and Whitman 2003). In the contiguous US, the largest populations are found in Montana and Idaho.





**Figure A3.2.** Current known and possible distribution of wolverines in North America (Copeland and Whitman 2003)

(Table A3.1, Figure A3.3), however, these are unverified reports. In 2006, the Institute for Wildlife Studies conducted a survey for wolverines in SEKI and the surrounding area and did not find any evidence of wolverine presence in the area. If wolverines are present in the parks, the IWS analysis indicates that the numbers are likely to be too small to constitute a viable population (Garcelon et al. 2009, Green 2007, Zielinski et al 2004).

**Table A3.1.** Wolverine observations in SEKI from 1925-1948 (SEKI 2010)

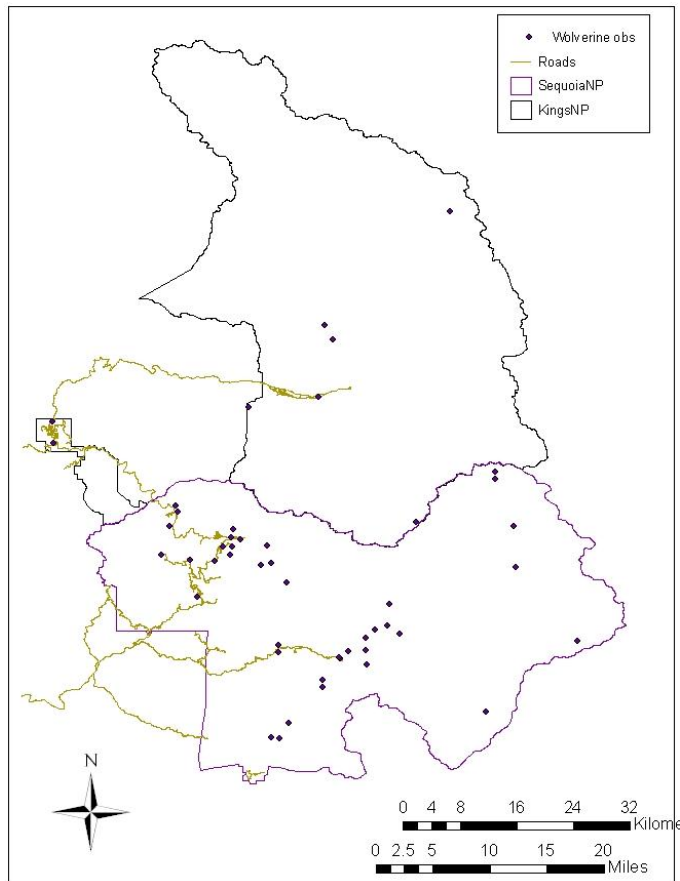
1925	4 wolverines trapped in the vicinity of Mineral King.
1939	2 juvenile wolverines observed near Little Baldy Saddle.
1941	Tracks observed fairly regularly above 7,000 ft. One line of tracks was observed on the Atwell-Hockett Trail near Deer Creek
1946	Tracks observed near Panther Gap during the winter
1947	Wolverine very rare at Three Rivers, CA. None were reported for several years
1948	A blurred track, believed to be wolverine found at Frypan on May 12, One track found at the head of Comb Creek on May 20

*California.* Wolverines are believed to have been extirpated from California by trapping in the early 1920's, though continuing credible reports in the Sierras suggest that some individuals may have survived (SEKI 2010). While trappers did not specifically target wolverines because of their low density, trapping lines may have attracted wolverines to feed on other trapped animals (Johnson 1988-1990). Poisoning was also likely to be a factor in contributing to declines (Garcelon et al. 2009). A male wolverine recently detected in the Lake Tahoe area was determined to be a long-distance immigrant from a Rocky Mountain population (Moriarty et al. 2009).

#### Sequoia and Kings Canyon.

According to park records, wolverines and wolverine tracks have been observed sporadically in SEKI until as recently as 2008





**Figure A3.3.** Reports of wolverine in SEKI from 1903-2008 (SEKI 2010)

### ***Conservation Status***

As a species, the wolverine is considered globally secure by NatureServe (G4 ranking) and vulnerable by the IUCN. Wolverines were a candidate species for listing as a federally endangered species in the US and have protected status in many states, including Washington, Oregon, Colorado, Idaho and Wyoming (Banci 1994, USFWS 2008). Wolverines are listed as threatened in the state of California (CDFG 2008).

### ***Threats and trends***

Wolverines are highly sensitive to human disturbance. In particular, logging activities and human recreation, such as backcountry skiing and snowmobiling, have been shown to negatively affect wolverine presence in otherwise undisturbed areas (Krebs et al. 2007). Trapping is a chief cause of mortality in populations where it is permitted and trapping mortality is generally additive, rather than compensatory (Krebs et al. 2004). Ultimately, the persistence of wolverines will likely depend on the availability of large tracts of land with minimal disturbance from humans (Copeland and Whitman 2003).

The availability of large ungulate prey species, such as mule deer, may also be important to sustaining wolverine populations (Garcelon et al. 2009). Management practices that reduce ungulate populations, such as excessive hunter harvest or loss of ungulate

wintering habitat, may negatively affect wolverine survival (Banci 1994). Alternatively, practices that improve ungulate habitat and the availability of carrion, may positively affect survival rates (Copeland and Whitman 2003).

Habitat fragmentation represents a significant threat to the persistence and re-establishment of wolverine populations. As alpine areas become progressively more fragmented in the landscape, opportunities for natural re-colonization and population re-establishment become increasingly limited. As a result, small and isolated populations are under increased threat of local extinction. Hunted populations may also experience declines if they are not supplemented by immigrants from undisturbed populations (Krebs et al. 2004). Finally, increased fragmentation of habitat may result in reduced gene flow, as evidenced by genetic sub-structuring of populations in Montana (Schwartz et al. 2009).

Snow cover that persists throughout the spring denning season is vital to successful reproduction (Magoun and Copeland 1998) and spring snow cover is the habitat variable that most closely accounts for historical distributions of wolverines in the contiguous United States (Aubry et al. 2007). Projected reductions in spring and winter snow cover due to climate change will likely have a detrimental effect on survival and limit available habitat for reproduction (Brodie and Post 2010). By reducing snow cover, climate change is likely to contribute to the growing isolation and fragmentation of populations, particularly in the southern portion of the species range.

### ***Conservation in Sequoia – Kings Canyon***

With the exception of a recent migrant detected in the Lake Tahoe region, studies have found no evidence that wolverines are currently living in the Sierra Nevada range (Garcelon et al. 2009, Moriarty et al. 2009). Thus, conservation actions targeting existing populations are unlikely to be fruitful. However, a recent study by the Institute for Wildlife Studies examined the feasibility of translocating wolverines to the Sierras in order to reintroduce individuals or augment the (potentially) existing population (Garcelon et al. 2009). This study determined that such a program is feasible due to the availability of high quality wolverine habitat under federal management and sufficiently high populations of ungulate prey species in the region. If translocations of wolverines are successful in establishing a viable population in the Sierras, SEKI will likely be important to maintaining that population, due to the large areas of protected habitat within its borders. Wolverines are highly mobile, so managing wolverine habitat both within and outside of parks boundaries would be necessary for establishment and continued persistence in the region.

The effects of climate change are likely to negatively affect the availability of suitable habitat for wolverines in the Sierras and within SEKI. In particular, reduced snow cover will likely have a detrimental effect on reproduction and limit the potential range of wolverines in SEKI.

### ***Data Gaps & Research Priorities***

Recent surveys have not been able to detect any wolverines present in SEKI or the surrounding region. Future research should continue to focus on detecting individuals in

the region. If a translocation program is established, research efforts should prioritize monitoring the seasonal habitat use, diet, and vital rates of translocated individuals to determine the efficacy of translocation protocols and the need for continued population augmentation.

### ***Summary and Recommendations***

The wolverine is a carnivorous mammal found at low densities in boreal and mountainous regions in North America and Eurasia. Wolverine populations were believed to be extirpated from California by 1922, although continuing credible reports in the Sierras suggest that some individuals may have survived. Wolverines are listed as threatened in the state of California. Recent surveys in Sequoia/Kings Canyon National parks indicate that wolverines are not likely to be present in SEKI (Garcelon et al. 2009).

In the absence of verified individuals in the region, no specific management actions are required. However, continued surveys targeting wolverines and other predators in SEKI will allow any existing individuals or new migrants to be detected. If a translocation program is undertaken, then direct management, additional monitoring and continued augmentation from outside populations will likely be necessary to sustain wolverines within and near SEKI. In particular, management actions to maintain connectivity of high quality alpine habitat in SEKI with undisturbed habitat in surrounding regions will be critical to the establishment of a viable population. Management of ungulate populations and limits on backcountry recreation will also improve the availability of resources and reduce disturbance from humans.

### ***Data Sources***

Species life history information was taken from NatureServe species accounts, California Department of Fish and Game reports, Copeland and Whitman (2003) and the primary literature. Distribution data was acquired for North America, California and Sequoia – Kings Canyon National Parks from NatureServe, Copeland and Whitman (2003) and the SEKI observation database. An evaluation of the current condition of the wolverine and threats to the species was compiled using US Fish and Wildlife Service and California Department of Fish and Game reports and the primary literature.

## A4. California Condor (*Gymnogyps californianus*)

### **Abstract**

The California condor (*Gymnogyps californianus*) does not currently forage or nest within the Sequoia and Kings Canyon National Parks, but could feasibly become a regular visitor if birds from nearby reintroduction programs continue to expand their range.

- **Species global status:** G1 (Globally endangered)
- **Park status:** Extirpated
- **Overall integrity:** Low
- **Certainty on integrity:** High (population size, habitat use, and location of individuals known)
- **Metric to evaluate integrity:** Population size (extirpated from park, globally rare, and dependent on human intervention to survive in the wild)
- **Vulnerability with respect to key stressors:** The California condor is at risk across its range from loss of food resources to development and contamination of those food resources by lead bullets. Sensitivity to climate change and air pollution is unknown.

### **Species Background**

The California condor was formerly a wide-ranging carrion-feeding bird. The condor is native to the Western US - including California and the area surrounding Sequoia and Kings Canyon National Parks. It is a member of the family *Cathartidae*, and is the largest North American land bird. Condors are long-lived with a relatively low reproductive rate. Most birds take 6-8 years to reach reproductive maturity, and will then lay only one egg every other year (Meretsky et al. 2000). The condor requires nest sites in a hollow or crevice on a cliff or steep slope, locations that deter golden eagle (*Aquila chrysaetos*) and common raven (*Corvus corax*) predation.

Condors require open areas where carrion (or the activity of other scavengers) can be easily located by sight. They are known to consume a wide variety of carrion, including large land mammals such as tule elk (*Cervus canadensis ssp. nannodes*) and mule deer (*Odocoileus hemionus*) as well as marine sources such as beached whales and elephant seals, which were historically found in abundance in California (Chamberlain et al. 2005). With European settlement and the implementation of cattle ranching in California in the 1700s and 1800s, food sources available to the California condor shifted to cattle carcasses (Chamberlain et al. 2005). Currently condors feed primarily on carcasses provided by land managers, though they do occasionally feed on animal remains discarded by hunters.

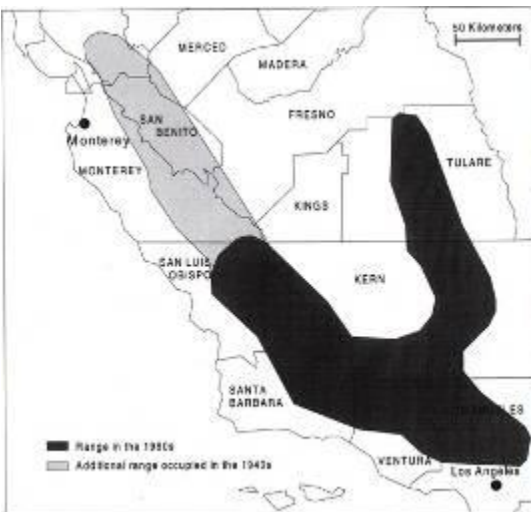
### ***Distribution and population trends***

**Global and Regional.** California condors historically ranged throughout the western United States (Figure A4.1) and fossil evidence from Florida and New York suggests they may have once been even more widespread (Snyder and Schmitt 2002). Over the course of 100 years (1880's- 1980's), the population and its range steadily declined to only 22 individuals in California (Figure A4.2). All remaining birds were taken into captivity at that time. The current Condor distribution is limited to three major reintroduction sites: (1) Reserves in Ventura, Santa Barbara, Kern, Monterey and San Luis Obispo counties in California, (2) Vermilion cliffs and Grand Canyon in Arizona, and (3) a remote area of Baja California, Mexico.



**Figure A4.1.** Historic and current California Condor distribution (Snyder and Schmitt 2002)

As of October 31, 2010, there were a total of 381 Condors, with 189 in captivity and 192 in the wild. Of the 192 condors in the wild, 94 are in California, 76 in Arizona, and 22 in Baja California, Mexico (USFW 2010). A population model, based on the wild population structure, predicted future releases, breeding effort, breeding success and mortality rates, projects that the California condor population will continue to grow slowly through 2014 (USFW 2009).



**Figure A4.2.** California Condor range in the 20<sup>th</sup> century. (Ventana Wildlife Society)

**Sequoia and Kings Canyon.** Sequoia and Kings Canyon National Parks fell within the reduced range of California condors in the 20<sup>th</sup> century (Figure A4.2). Condors have historically been observed in Sequoia and Kings Canyon on occasion, with just 41 observations recorded in the parks' observation database (SEKI 2010). Recorded observations began in 1907 and ended in 1981, just a few years before all remaining wild condors were taken into captivity. Historically condors may have used the Sequoia and Kings Canyon area for nesting. Active nest sites in Sequoia trees were occasionally documented (Snyder and Snyder 2000).

While condors are not currently found foraging or nesting in Sequoia and Kings Canyon, they may begin appearing in the park in the coming years as nearby reintroduced populations expand. The nearest release site is at the Bitter Creek National Wildlife

Refuge in Kern County, and birds from this site have ranged as far as the foothills east of Bakersfield, but not yet as far north as Sequoia and Kings Canyon (pers. comm. George 2010, pers. comm. Werner 2010). Historic reports suggest that condors visiting the park generally fed on dead cattle outside the park boundaries. Ranches to the west of the park could serve as a supplemental food source for any reintroduced condors. However, pending development in these areas would reduce such potential in the future, further limiting food availability near the parks (pers. comm. Werner 2010). Thus, any reintroduction of condors to the park would likely require a food source that does not currently exist within the park.

### ***Conservation concern***

Today California condors are an intensively managed endangered species. Successful breeding among released condors is rare due to behavioral issues such as preferential feeding of trash, including small non-digestible metals and plastics, to young. There is a particular concern over lead ingestion through lead shot used for hunting. Lead poisoning from hunting continues to severely impact adult mortality (Church et al. 2006, Cade 2007). Trash feeding of chicks may be related to both availability of trash and behavior (Walters et al. 2010). Condors appear to require small amounts of bone to fulfill nutrient requirements unmet by ingestion of carrion muscle and organs – and it is possible that condors mistake small pieces of plastic and metallic trash for these nutrient-rich bone fragments (Walters et al. 2010). Lead poisoning has required numerous interventions by condor biologists in the form of chelation therapy (Walters et al. 2010). However, lead-free bullets are increasingly used in condor foraging territory and if this trend increases, the need for emergency intervention for lead-poisoned birds could be virtually eliminated (Walters et al. 2010).

### ***Continued stewardship***

Due to approximately 30 years of intensive captive breeding and reintroduction programs, the natural history and threats to the California condor are relatively well known. However, the species vulnerability to climate change has not been specifically assessed to our knowledge. Climate change could alter the availability of carrion or have effects on condor physiology or disease incidence, but the likelihood of these scenarios is unknown. While proximate threats to the recovery of the California condor (such as lead bullets) remain, climate change is likely a secondary threat to species persistence and population expansion (Walters et al. 2010). However, future research should focus on how climate might affect the species capacity to thrive in the absence of human intervention as this might affect the choice of reintroduction sites.

## A5. Brown Bear (*Ursus arctos*)

### **Abstract**

The brown bear (*Ursus arctos*) is extinct in California, but globally secure due its population in northern latitude locations of North America While historically threatened by hunting, habitat loss and conflict with humans are current threats to the brown bear.

- **Species global status:** G4, SX (CA) (Globally secure, extinct in California)
- **Park status:** Non-extant
- **Overall integrity:** Low
- **Certainty on integrity:** High
- **Metric to evaluate integrity:** Population size in California (none in the wild).
- **Vulnerability with respect to key stressors:** Extirpated due to hunting. If reintroduced would be vulnerable to human-wildlife conflicts.

### **Species Background**

**Life History.** The brown bear (*Ursus arctos*) is a large omnivorous bear with no non-human predators. In the United States, their diet consists of a wide variety of food items including fish, berries, fungi, rodents, carrion, insects, roots, honey, small ungulates, and tender herbaceous plants. Although they are not true hibernators, they do den throughout much of the winter in a heavy sleep and need to gain large fat reserves in the fall to survive winters. They are largely solitary; however, they will congregate around abundant food sources such as salmon runs. The replacement rate of *U. arctos* is relatively slow. Females are sexually mature around 5-7 years, and generally give birth to 1-4 cubs (usually 2) every few years. The cubs stay with their mother for 2-4 years (Wilson & Reeder 2005).

This bear is an iconic symbol for the State of California even though it was extirpated from the state in 1922. It is featured on the state flag and has cultural and historical importance in California.

**Taxonomy.** The genetic diversity and subspecies of brown bears is in debate, with various DNA analyses revising earlier phylogenies. Although some sources propose dozens of sub-species (Grinnell 1937), recent DNA analyses support 16 sub-species world-wide (Waits et al. 1998). Only one of these sub-species (*U. arctos horribilis*) currently occurs in the United States. The sub-species that likely inhabited the area near Sequoia and Kings Canyon National Parks (*U. arctos californicus*) is extinct (Wilson and Reeder 2005).

### ***Distribution and Population Trends***

*Global and Regional Context.* *Ursus arctos* has a broad global distribution including North America, Asia, and Europe (Figure A5.1). It has experienced great range reductions throughout much of its range worldwide. Although this species was once abundant throughout the western half of the lower 48 states of the United States (Figure A5.2), there currently exist only five small populations in that area. Most of the individuals persisting in the lower 48 states occur in protected areas with large tracts of un-fragmented habitat (e.g., North Cascades National Park and Yellowstone National Park).

Local extinctions of grizzly bears in the United States have been a product of a variety of threats including: habitat destruction (mainly through development and agriculture), hunting (for sport, protection, to protect livestock), and loss of key food resources (such as dwarf elk in Sutter Basin) (Grinnell 1937). These bears can threaten human livelihoods by killing livestock. While incidents are infrequent, they have also been known to attack people (Grinnell 1937). Because of their relatively low tolerance for human disturbance, and the threat they represent to humans, brown bear populations in the lower 48 states are most likely to thrive in protected areas such as Yellowstone National Park (Schwartz et al. 2002).

Populations of *U. arctos* in Alaska and the Northern Rockies have survived hunting and habitat pressures and are increasing or stable in some areas. Yellowstone National Park has seen a recent increase in population size with a 48% increase in range from the 1970's (when brown bears were federally listed in the lower 48 states of the U.S.) to the 1990's (Schwartz et al. 2002). Canada and Alaska harbor substantial populations (~97% of brown bears in North America reside in Alaska or Canada).

*Statewide and Local Distribution.* In California, brown bear populations were exterminated by hunting during the 19<sup>th</sup> and 20<sup>th</sup> century. During early European settlement, these bears were so abundant in western and central California that they were considered a great nuisance and safety threat. Although there is little reliable information concerning *U. arctos* in California from early European settlement, the populations here seemed to be restricted to lower elevations (Grinnell 1937). There are numerous historic accounts of these bears throughout the central valley and coastal areas, but relatively few accounts in the foothills and Sierra Nevada Mountains (Grinnell 1937).

The brown bear occurrences documented near Sequoia-Kings Canyon National Parks may have been the last brown bears confirmed in California. One bear was shot near the park in 1922 (last confirmed brown bear in California). There were also a series of brown bear sightings in Sequoia-Kings Canyon area in 1924 but these latter sightings may have been cinnamon-colored black bears (Grinnell 1937).

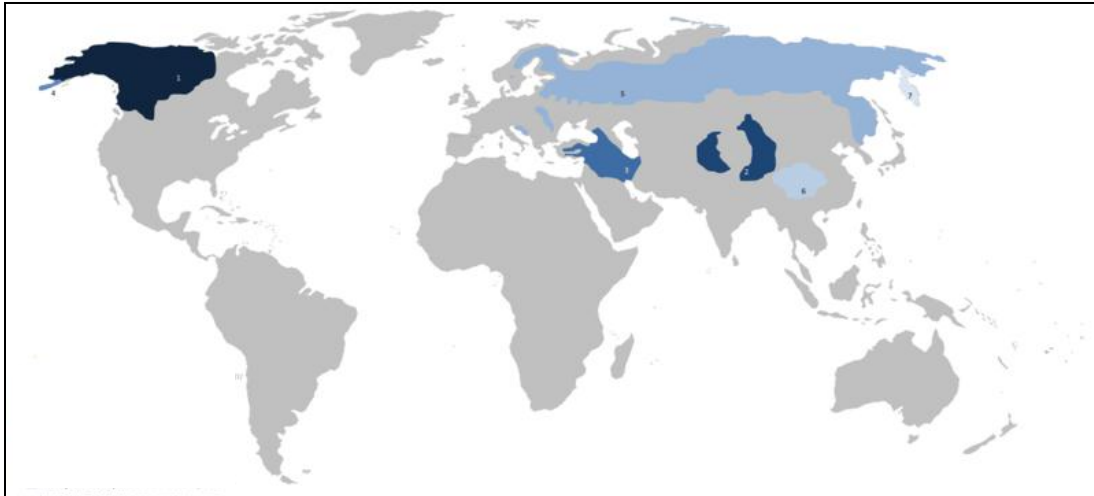


### **Conservation Status**

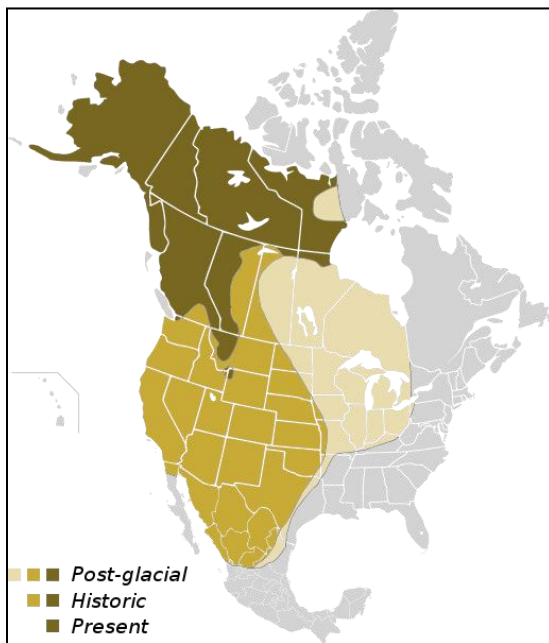
Globally, *U. arctos* is in decline. It has been extirpated from much of its range, but persists in large numbers in a few areas such as Alaska, Canada, and Eastern Asia. The Brown Bear Working Group has designated the following areas as core areas for conservation priority: Alaska, republics of the Russian Far East, and Hokkaido, Japan (Brown Bear Working Group 2007). Globally, the species is considered a Species of Least Concern by the IUCN (McClellan et al. 2008). However, population segments within the lower 48 states were federally listed under the Endangered Species Act as threatened by the US Fish and Wildlife Service in 1975 (Endangered Species Act 1975).

### **Management potential in Sequoia Kings Canyon National Park**

Brown bears are currently not managed in the park because they have been extirpated. However, it is theoretically possible to introduce brown bears from other areas to the park. There are several observations that argue against this proposition. First, a reintroduction would consist of novel genotypes since the native subspecies is extinct (Brown Bear Working Group, 2007). While a novel sub-species may succeed, the rationale to support this decision would rest on restoring ecosystem functioning lost through the loss of brown bear. There is no compelling argument regarding what these lost functions are.. Second, the park habitats were historically not used as permanent territories. It is likely that the brown bears that once lived in the park were seasonal. It may be that human activity, reducing habitat in the Central Valley forced the bears into montane habitats during the last years of their existence in California (Ginnell 1937; Werner, personal communication). Thus, our best ecological evidence suggests that the Parks do not provide good habitat for brown bears. Third, park food resources for brown bear are likely inadequate. Although some salmon are present in the park, the park likely does not provide a large food source for brown bears in the fall when they are gaining weight for winter. In Yellowstone National Park the brown bears eat a large amount of small rodents and berries in the fall which are not abundant in Sequoia/Kings Canyon Parks (Schwartz et al., 2002; Werner, personal communication). Finally, the amount of space needed for a healthy population is likely much larger than these parks themselves and would require the cooperation of surrounding jurisdictions (Brown Bear Working Group, 2007). Introduction of brown bears would likely result in human-wildlife conflict as a consequence of the need for large territories, and the generalist nature of bear diet. For these reasons, introduction of brown bears into the park would likely not be successful.



**Figure A5.1** Current global distribution of *Ursus arctus* separated by subspecies.



**Figure A5.2** Former and present distribution of *Ursus arctos* in North America.

## A.6. Sierra Nevada Bighorn Sheep - *Ovis canadensis sierrae*

### Abstract

- **Species global status:** G4T1 – Species is apparently secure; subspecies is critically imperiled and is listed as endangered by USFWS (NatureServe 2010).
- **Park status:** Rare, patchy – Occur only in limited areas of the parks.
- **Overall integrity:** Low, but improving – As compared to pre-settlement era, population size is low (USFWS 2007), but has been increasing over the last decade (Wehausen et al. 2009).
- **Certainty on integrity:** High - As a highly managed and monitored species, estimates of the status and trends of bighorn populations are relatively certain.
- **Metric to evaluate integrity:** Population size
- **Vulnerability with respect to key stressors:** Bighorn populations are susceptible to disease, climate change and, to a lesser extent, altered fire regimes (see discussion below).

### Species background

**Taxonomy.** Sierra Nevada bighorn sheep (*Ovis canadensis sierrae*) is a mammal of the order *Artiodactyla* and family *Bovidae* (NatureServe 2010). The group was first described as a unique subspecies by Grinnell in 1912, following Sierra Club sightings near the headwaters of the Kings and Kaweah rivers in 1911, and sightings by cattlemen in Inyo County. Four sheep from the Mount Baxter herd were taken as specimens, and used for Grinnell's description (1912). Prior to the 1911 sightings it was believed that bighorn sheep had been extirpated from the Sierra Nevada.

Thirty years later, Sierra bighorn were reclassified as belonging to the subspecies *O. canadensis californiana*, which included the bighorn ranging from the Sierra Nevada north into British Columbia (Wehausen et al. 2005, CDFG 2007). Wehausen and Ramey (2000) refuted this umbrella classification using cranial morphometric analysis. The resulting reclassification put bighorn sheep in the southern and central Sierra Nevada in a unique group (*O. c. sierrae*). The populations from British Columbia and Washington were designated as *O. c. canadensis*, and those from Oregon and southwestern Idaho as *O. c. nelsoni*. These results were further supported by mitochondrial DNA analysis (Ramey 1995). Thus, Sierra Nevada bighorn sheep are now regarded as a unique subspecies, with *O. c. sierrae* being considered the most appropriate name.

**Life history.** Sierra Nevada bighorn sheep are large ungulates that live along the Sierra Nevada crest and on the range's eastern slopes. They are seasonal migrants, but consistently stay on or near rocky open areas. The bighorn rely on vision to detect predators, and escape predation by fleeing to steep, rocky areas where they are able to move with agility (USFWS 2007). Since vegetation for grazing is limited in rocky areas, bighorn tend to forage in open steppe vegetation with accessible precipitous areas that can serve as escape terrain (Wehausen 1980). Bighorn sheep further decrease predation risk by living in groups (USFWS 2007).

Bighorn sheep are polygynous, with socially dominant males mating with the greatest number of females. Males and females typically live separately, but come together to

breed in late fall and generally remain in mixed sex groups during the winter (Wehausen 1980, USFWS 2007). Ewes typically live longer and are more numerous than rams. *O. c. sierrae* have significantly shorter life spans than other North American wild sheep, with eight years being considered an old age for a ram (Wehausen 1980).

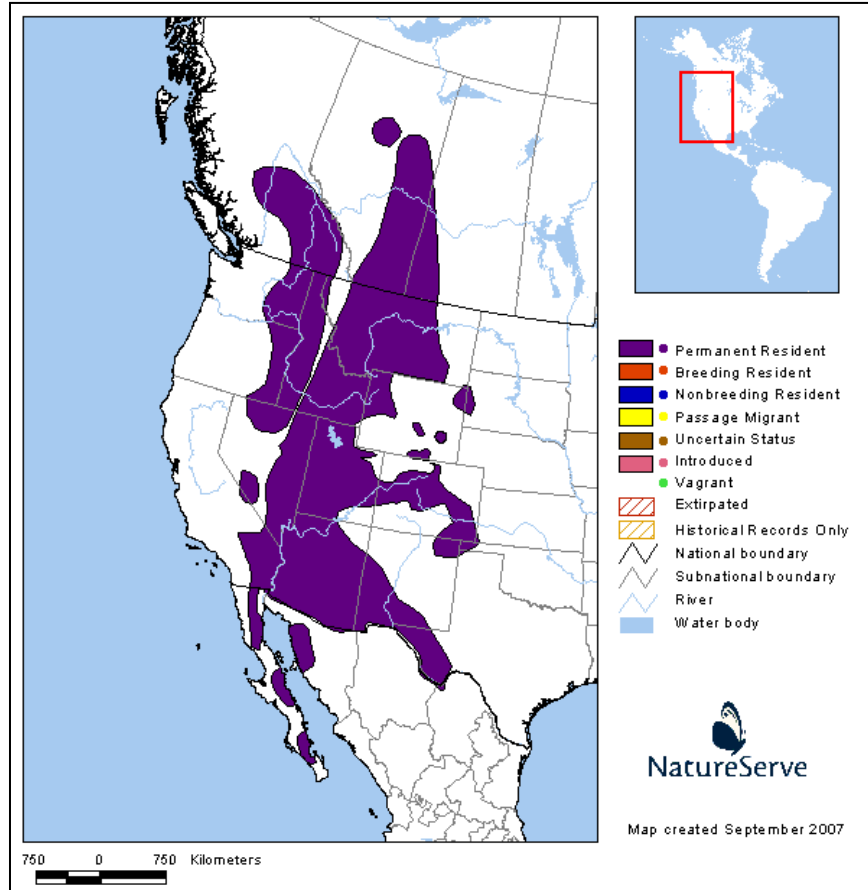
Foraging habitat varies seasonally. In winter, when nutrient quality of their diet is lowest, bighorn prefer south facing slopes, where early fall snowmelt initiates growth of grasses such as needlegrass (*Stipa speciosa*). Bighorn may use a winter forage site for several days before moving on. In spring they transition to eating newly grown shrubs, and will move lower in elevation if habitat is sufficiently open and close to rocky areas. As snow melts, rams tend to shift upward in elevation coincident with the zone of maximum vegetative growth, and generally stay below timberline. In contrast, ewes move to higher elevation where the foraging quality is lower, but the steep terrain provides relative safety for lambing. Ewes and lambs remain in alpine habitat until lambs are weaned in the fall. During the summer, meadows and brush patches are important forage areas. Groups of ewe and lamb move frequently between these small and scarce forage patches. Nutritional value of bighorn diet increases throughout the spring, then declines as fall approaches, when it may limit population growth (Wehausen 1980). Bighorn sheep procure most of their water needs from their food (USFWS 2007).

Bighorn tend to choose the safety of open rocky sites over more nutritionally advantageous areas. For example, Wehausen (1996) documented bighorn sheep abandoning their lower elevation winter ranges to avoid predation by mountain lions, despite a reduction in nutrition availability and decreased population growth. In their winter range, when snowy conditions force bighorns to move to lower elevation, they choose barren areas with intermediate amounts of available forage. Richer foraging areas such as forests and areas with large shrubs are avoided due to their limited ability to detect predators in dense vegetation (CDFG 2008c). This choice for safety over forage quality is also evidenced by ewes in spring, which migrate out of nutritionally optimal lower elevations to alpine environments for lambing (Wehausen 1980).

Lambing occurs in spring, typically between late April and early June, and lasts for just under a month. Weaning occurs around the beginning of October. Milk quantity and quality are influenced by both pre- and post-natal nutrition (Wehausen 1980). Onset of lambing is delayed compared to other bighorn subspecies; Sierra Nevada bighorn ewes typically begin lambing at age 3, although larger individuals may begin at age two. Lambs are generally born singly, but twins can occur under favorable nutritional conditions (Wehausen 1980). Male lambs stay with ewes until they are around two years old, at which point they adopt the lifestyle of rams.

## Distribution

**Global.** Bighorn sheep (*O. canadensis*) are found throughout western North America (NatureServe 2010, Figure A6.1) Sierra Nevada bighorn sheep (*O. Canadensis sierrae*) are found only within the Sierra Nevada.



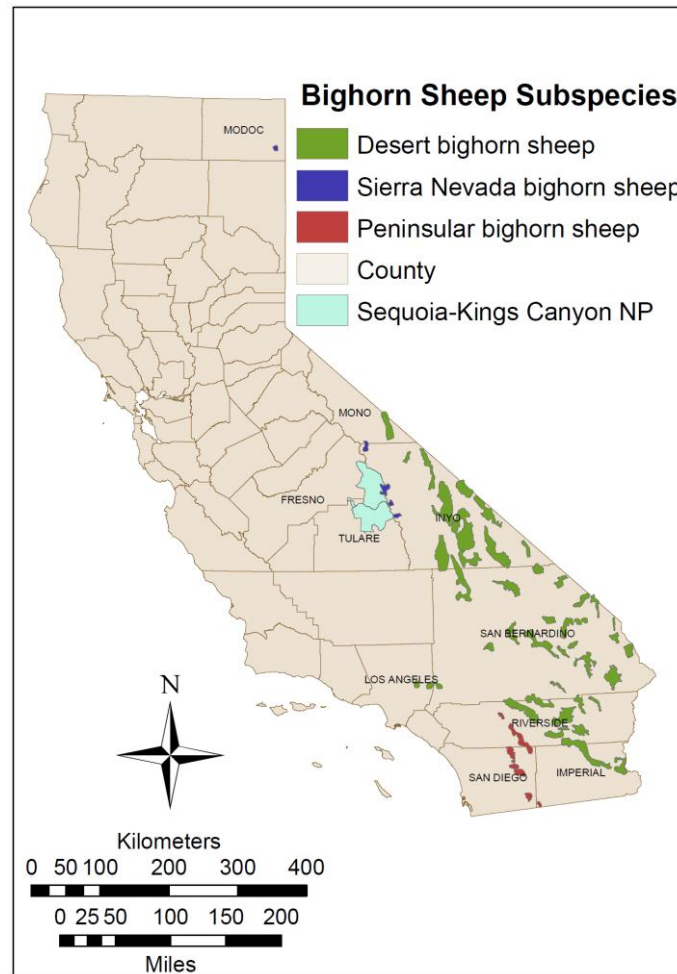
**Figure A6.1.** Distribution of all subspecies of *Ovis canadensis*. Data provided by NatureServe in collaboration with Bruce Patterson, Wes Sechrest, Marcelo Tognelli, Gerardo Ceballos, The Nature Conservancy-Migratory Bird Program, Conservation International-CABS, World Wildlife Fund-US, and Environment Canada-WILDSpace.

**Regional.** Recorded observations from naturalists such as John Muir as well as recovered bighorn remains indicate that historically, bighorn sheep had a wide range in the Sierra Nevada from Olancha in the south to Sonora Pass in the north and Mineral King area to the west (CDFG 2007). Table A6.1 and Figure A6.2 summarize current locations within the state of California.

**Table A6.1.** Summary of presence of the three bighorn sheep subspecies found in California counties.

	Fresno	Imperial	Inyo	Los Angeles	Modoc	Mono	Riverside	San Bernardino	San Diego	Tulare
Desert		•	•	•		•	•	•		
Peninsular		•					•		•	
Sierra Nevada	•		•		•†	•				•

• Natural heritage records exist for county  
†Extirpated/possibly extirpated



**Figure A6.2.** Depiction of the California Natural Diversity Database (CNDDB) distribution data of the three bighorn sheep subspecies found within California. Note that following reintroduction, the Modoc herd unit was extirpated due to disease transmission from domestic sheep (CDFG 1990).

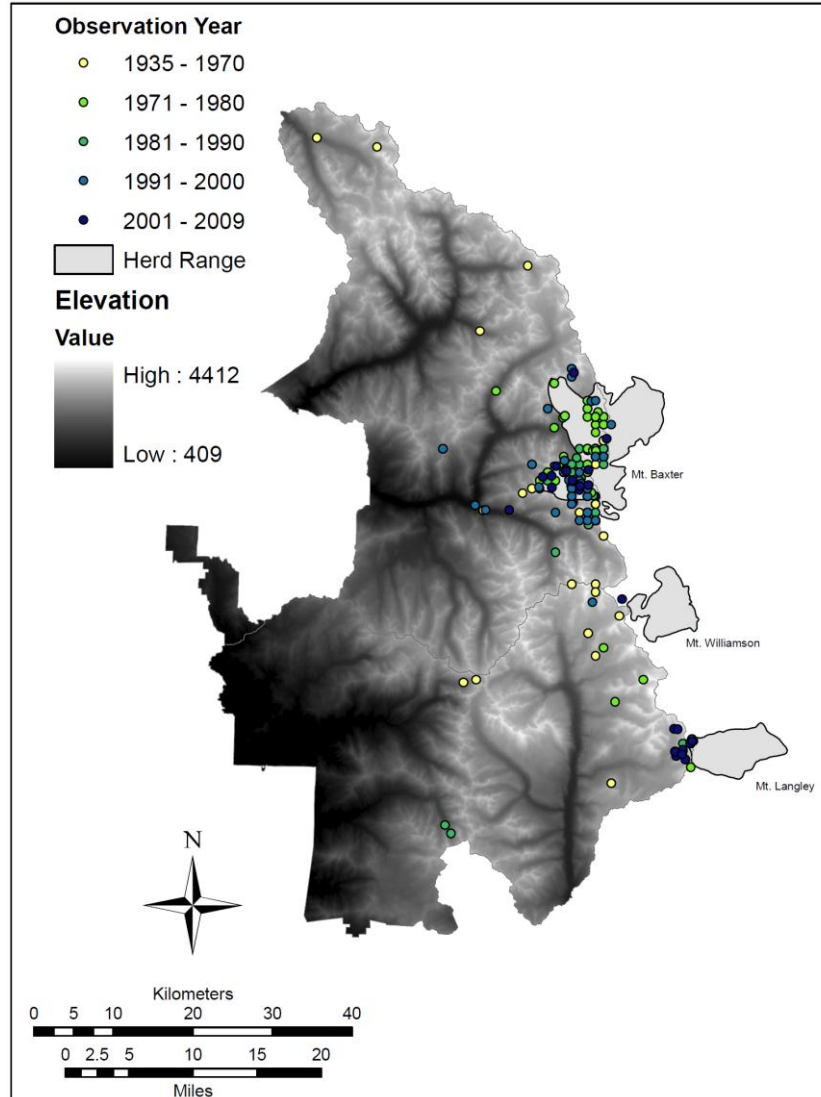
*Sequoia Kings Canyon National Parks.* Six herds of Sierra Nevada bighorn sheep occur along the border of Sequoia and Kings Canyon national parks and the Inyo National Forest. These are the Wheeler Ridge, Mt. Baxter, Sawmill Canyon, Bubbs Creek, Mt. Williamson and Mt. Langley herds. The ranges of the Mt. Baxter and Sawmill Canyon herds include terrain in both Kings Canyon National Park and the John Muir Wilderness area, while that of the Bubbs Creek herd is entirely within the park. Wheeler Ridge, Mt. Williamson and Mt. Langley herds are primarily outside of the park, although bighorn sheep observations indicate individuals will stray across the park boundary ( Figure A6.3).

### **Condition and Trends**

The Sierra Nevada bighorn sheep has a conservation rank of G4T1 (meaning the species is apparently secure globally, but the subspecies is critically endangered) and is listed as an endangered species by the US Fish and Wildlife Service (USFWS) (NatureServe 2010). Conservation status is further discussed in the management section below. In recent years the total Sierra Nevada bighorn sheep population appears to have increased to around 400 individuals distributed over eight herd units (Table A6.2, Figure A6.3).

**Table A6.2.** Summary of population data as of summer 2008. Herds relevant to Sequoia and Kings Canyon national parks are in bold. All values are minimum numbers present except where marked with an asterisk, which include mark – resight estimates. Table was adapted from Wehausen et al. 2009.

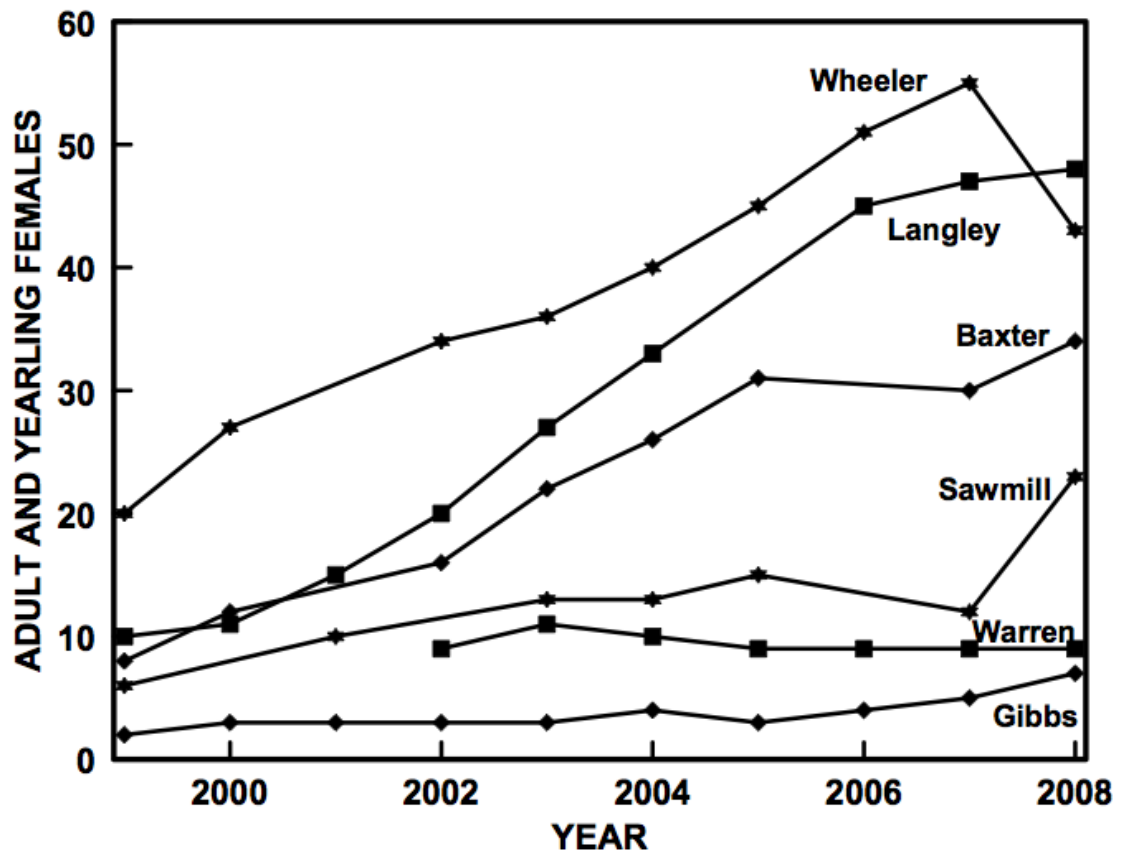
HERD UNIT	ADULT EWES	YRLG EWES	TOTAL EWES	LAMBS	ADULT RAMS	YRLG RAMS	TOTAL RAMS	TOTAL
Langley	35	3	48*	8	19	5	24	80*
Williamson	11	3	14	4	8	2	10	28
Bubbs	14	3	17	1	4	1	5	23
Baxter	29	5	34	13	12	5	17	64
Sawmill	22	1	23	9	8	3	11	43
Wheeler	36	2	43*	14	20	2	31*	88*
Gibbs	5	2	7	3	3	1	4	14
Warren	6	2	8	5	7	7	7	21
Total	158	21	194*	57	81	19	111*	362*



**Figure A6.3.** Sequoia and Kings Canyon observations of bighorn sheep between 1935 and 2009. Ranges of the three relevant Sierra Nevada herds are also shown (Mt. Baxter, Bubbs Creek and Sawmill Canyon considered one herd here), covering both National Park and Inyo National Forest. Wheeler Ridge population in the far northern portion of the park is not mapped.

*Mt. Baxter, Bubbs Creek & Sawmill Canyon Herds.* After increases earlier this decade, the Mt. Baxter herd population appears to have leveled off in recent years. The Sawmill Canyon herd has remained relatively stable over the past decade, but has shown an increase in population from 2007-2008 (Figure A6.4). Mortality rates due to mountain lion predation are likely limiting population growth in both herds (Wehausen et al. 2008, 2009). The Bubbs Creek herd saw a population increase to 28 sheep in 2008, and maintained that size in 2009 (Wehausen et al. 2009).





**Figure A6.4.** Recent reproductive base histories for 6 herd units of bighorn sheep in the Sierra Nevada for which adequate data exist. Data are all minimum counts except for mark-resight estimates for the Mt. Langley herd unit after 2006 and the Wheeler Ridge herd unit after 2005. Figure from Wehausen et al. 2009.

*Mt. Williamson Herd.* It has been difficult to obtain a complete count of the Mt. Williamson herd following a shift in winter ranges after 1985. An uncommon opportunity for a good direct count occurred in the summer of 2007 where 28 different sheep were observed (Wehausen et al. 2008). In 2008 the population was again counted as 28 sheep (Wehausen et al. 2009). This minimum population is similar to good winter counts between 1978 and 1985 (ranging from 29 to 31 total sheep), indicating that at a minimum, the Mt. Williamson herd population is stable (Wehausen et al. 2008). Similarly to the Mt. Baxter and Sawmill Canyon herds, mountain lion predation has been implicated as a curb to population growth (Wehausen et al. 2009).

*Mt. Langley Herd.* Similarly to the Mt. Baxter herd, growth of the Mt. Langley herd unit has slowed in recent years, following strong increases in previous years (Figure A6.4). The population estimate for 2008 was only 80 individuals (Table A6.2), down from approximately 100 in 2006 (Wehausen et al. 2008). Observations in earlier years indicated that the herd population may have been approaching carrying capacity. In October of 2008 an ewe carcass was found northeast of Crabtree Lake in Sequoia

National Park. This observation plus location data from a GPS collared bighorn female in the Crabtree Lakes area suggest that the Mt. Langley herd may be expanding its range westward into the national park (CDFG 2008c).

### ***Threats***

The greatest threats to Sierra Nevada bighorn sheep are exposure to disease from contact with domestic sheep, predation stress from mountain lions and increased vulnerability to stochastic events due to small population size and a lack of genetic diversity. However, habitat integrity and climate change may also play a role.

*Disease.* Bighorn sheep are known to be particularly susceptible to diseases carried by domestic sheep, especially pneumonia caused by *Pasteurella* infection. Pneumonia has been known to cause extirpations of entire populations of bighorn sheep in the United States. Disease transfer from domestic sheep is suspected to have played a major role in bighorn sheep declines in the late 19th century and while bighorn sheep and domestic sheep herds are deliberately segregated today, strays from domestic or wild herds remain a concern (USFWS 2007).

*Predation.* Predation pressure from mountain lions, and to a lesser extent bobcats (CDFG 2009b), has contributed to declines in Sierra Nevada bighorn sheep in two ways: 1) direct loss of individuals from depredation, and 2) a decrease in winter range use by bighorn herds which subsequently affects reproductive success negatively due to reduced forage and exposure to harsher winter conditions (Wehausen 1996, USFWS 2007).

*Small Population Size.* Sierra Nevada bighorn sheep have undergone periods of extremely low population size and continue to be comprised of only a few small herds. These bottleneck events have led to loss of genetic variation resulting in a reduced ability to adapt to threats such as disease and a changing environment. Likewise, small populations are vulnerable to random variation in population characteristics (e.g. sex ratio) and extreme environmental events such as avalanches and harsh winters (USFWS 2007). The Sierra Nevada subspecies is especially susceptible to these problems as its genetic diversity falls on the lower end of bighorn sheep populations sampled in the Southwest.

*Habitat Integrity.* The majority of the historic range of Sierra Nevada bighorn sheep falls within public lands and is not threatened by human development. There has been some deterioration of habitat quality due to fire suppression, with an increased number of trees in alpine habitats obscuring vision and limiting predator detection (CDFG 2008b). Illegal farming of marijuana also has the potential to negatively impact bighorn sheep. For example, in July 2007 marijuana groves were discovered on the winter ranges of the Mount Williamson herd (CDFG 2007). However, habitat loss and fragmentation are not generally considered major threats to Sierra populations. Likewise, poaching has not been documented in recent decades and is not considered a current threat (USFWS 2007).

*Climate Change.* While changes in temperature and precipitation are unlikely to directly affect Sierra Nevada bighorn sheep, alterations of alpine habitat due to climate change

have the potential to negatively impact bighorn populations. Sierra Nevada bighorn are dependent on open alpine habitat for predator avoidance and more specifically, alpine meadows as a source of valuable forage. Meadows are especially important during late summer and autumn when forage is sparse throughout much of the alpine zone. As the regional climate warms it is expected that the tree line will advance, reducing suitable alpine habitat. Additionally, warmer temperatures and the potential for precipitation to fall increasingly in the form of rain rather than snow will likely cause snowmelt-dependant alpine meadow vegetation to dry and senescence earlier in the growing season (Wehausen 1980, CDFG 2009a). With the potential loss of prime bighorn habitat and drying of meadows, Sierra Nevada populations are likely to suffer in the coming decades, although accurate projections of population change attributed to climate change are difficult to make.

### **Management**

*Regional.* Bighorn sheep in the Sierra Nevada were not well studied until the late 1970s. The California state legislature granted seasonal protection to bighorn sheep in 1876, and any take of bighorn sheep has been prohibited by the state since 1883 (USFWS 2007). However, this law did not address risks other than hunting, and numbers continued to fall (Andaloro and Ramey 1981). Further protection came in the 1930s when the US Forest Service (USFS) reduced grazing allotments for domestic sheep to lessen the negative effects of disease transmission and overgrazing on wild sheep populations.

In 1940 the Sierra Club proposed the establishment of a preserve for Sierra Nevada bighorn sheep, but it was declined on the basis that there was not sufficient justification at that time. Refuges were eventually created in 1971, with the involvement of both USFS and the National Park Service (NPS). Inyo National Forest designated 41,000 acres within the ranges of the two remaining herds (Mt. Williamson and Mt. Baxter/Sawmill Canyon) as California Bighorn Sheep Zoological Areas, while NPS disallowed off trail use in the western range of the Mt. Baxter herd (Wehausen 1980). These laws were later amended; in 1976 off-trail use in the herd's eastern range was allowed in summer below 3050 meters, and in 1977 NPS extended off-trail use east of the Rae Lakes drainage in the Baxter herd summer range, and established a one-night camping limit for the Baxter Lakes Basin (Wehausen 1980). The ban on off-trail use in Sequoia and Kings Canyon National Parks was lifted in 2001 (USFWS 2007).

Active management of the subspecies continued throughout the 1970s and 1980s. In 1972 the state designated *Ovis canadensis californicus* (synonymous with *O. Canadensis sierra*) as rare (USFWS 2007). Reintroductions began in the late 1970s, using the Mt. Baxter herd as a source population. Reintroductions were made to Wheeler Crest and Mt. Langley in 1979, 1980 and 1982 (Sierra Bighorn Interagency Advisory Group 1984). Introductions east of Mount Langley in the southern Sierra and in the Warner Mountains of Modoc County also occurred (Andaloro and Ramey 1981), and in 1986 the first reintroduction to the central Sierra was made, in Lee Vining Canyon (Keay 1987).

The state designation was changed from rare to threatened in 1984 (USFWS 2007), at which time there were fewer than 300 sheep remaining in the Sierra Nevada. A recovery plan for the species was drafted, with the goals of establishing at least three populations

of 100 or more, and restoring populations to all potential habitat within the historic range while maintaining “genetic integrity” (Sierra Bighorn Interagency Advisory Group 1984). As populations continued to dwindle through the 1990s and into the new century, concern for the subspecies led to more regulation. From 1985-1996 Sierra Nevada bighorn sheep were listed as a level 2 candidate for the Endangered Species Act, meaning that although listing was possibly warranted, sufficient information on the threat to the species was lacking. In 1997 CDFG drafted, and in 1999 implemented, a more intensive recovery plan that would involve population monitoring, actions aimed at reducing mortalities, continued reintroductions, and the possibility of a captive breeding program. Full listing under the Endangered Species act for Sierra Nevada bighorn sheep was granted in January 2000 in an emergency listing, at which time an estimated 125 adults remained (USFWS 2007).

Currently all herds in the Sierra Nevada are monitored for population trends and habitat use patterns using field counts as well as telemetry (Wehausen et al. 2008). The management strategy includes population augmentation, predator control, and habitat preservation. The most appropriate management strategy is herd specific. For example, preventing adult mortality by controlling predators was recently found to be more relevant to the Mt. Wheeler population, while introduction of more females would be more beneficial to the Mono Basin population (CDFG 2009b).

*Sequoia and Kings Canyon.* The National Park Service contributed founding members to the Sierra Nevada Bighorn Sheep Interagency Advisory Group, and has played an active role in creating and implementing management strategies, such as the Recovery and Conservation Plan of 1984, the USFWS Recovery Plan of 2007, and a 1986 management plan for sheep within the parks (USFWS 2007). NPS also provided funding for the instrumental research conducted by Wehausen in the 1970s (Wehausen 1980). There are currently six herds whose ranges fall at least partially within the boundary of Sequoia and Kings Canyon national parks: Wheeler Ridge, Mt. Langley, Mt. Williamson, Sawmill Canyon, Mt. Baxter and Bubbs Creek. Three of these, Mt. Williamson, Sawmill Canyon and Mt. Baxter, were the last remaining herds identified in the 1970s (Wehausen 1980). The parks contain additional suitable habitat that is currently unoccupied. Of the twelve essential herd units identified as necessary for recovery, eight overlap with Sequoia and Kings Canyon national parks (USFWS 2007, Figure A6.5). In addition to the herds identified above (with the exception of Bubbs Creek, which was not considered essential), these include Taboose Creek along the Sierran crest, as well as Laurel Creek and Big Arroyo west of the Kern River. Western expansion of the Mt. Langley herd into the Crabtree Lakes area was recently documented (CDFG 2008c), showing the potential for bighorn sheep to recolonize former habitat within the parks.

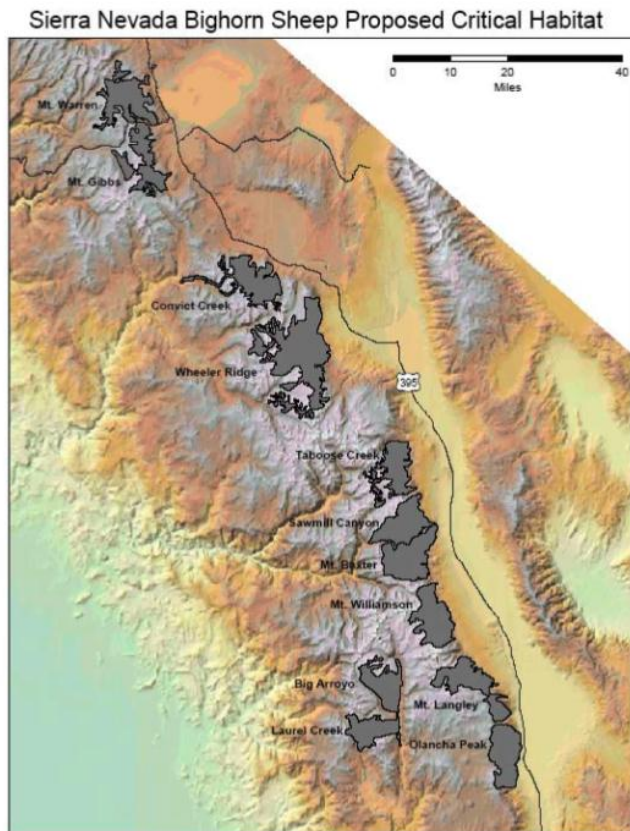
### **Data Gaps & Research Priorities**

Sierra Nevada Bighorn Sheep are a highly managed and researched species. Therefore, our understanding of the subspecies is more complete than many other threatened taxa and data gaps are less substantial. However, our understanding of how emerging threats such as climate change will affect the sheep is incomplete and further research into such areas are crucial for informing future management decisions.

### **Summary and Recommendations**

Populations of Sierra Nevada Bighorn Sheep were historically widespread across the Sierra Nevada range, but experienced dramatic declines following European settlement of California and introduction of domestic sheep. By the early 1900s it was believed that bighorn sheep were extirpated from the southern Sierra Nevada.

However, following government protections and increased management the subspecies has experienced some recovery. While the Sierra Nevada bighorn sheep remains a rare subspecies, most populations have seen increases over the last decade as a result of intensive management practices, including population augmentation, habitat conservation, and predator control. Decisions regarding continued implementation of such actions would be most effective on a herd to herd basis, with monitoring data informing management decisions. Much of the critical bighorn habitat identified in the USFWS recovery plan is in or adjacent to Sequoia and Kings Canyon National Parks. This includes the Mt. Baxter herd, which has been used as a source of translocations, as has the Wheeler Ridge population (1) two rams from Wheeler Ridge to Mt. Warren in 2005; 2) five ewes from Wheeler Ridge to Baxter/Sawmill in 2007; and, 3) three ewes from Wheeler Ridge to Lundy Canyon and 3 ewes from Mt. Langley to Lundy Canyon in 2009, D. Gammons, personal communication). Therefore it is important for park staff to remain involved in conservation efforts. The most prominent threats to the Sierra Nevada Bighorn Sheep included disease transmission from domestic livestock, high predation rates, small population size and possibly climate change. The most important thing park managers can do to help conserve the subspecies is to continue to promote and support ongoing monitoring and research of bighorn sheep in and around Sequoia and Kings Canyon national parks and support additional reintroductions at Big Arroyo and Laurel Creek, as required for delisting. If climate change leads to loss of alpine habitat,



**Figure A6.5.** Units proposed by the U.S. Fish and Wildlife Service as critical habitat for Sierra Nevada Bighorn Sheep. Figure from CDFG 2008a.

prevention of forest expansion into the alpine zone would benefit the Sierra Nevada Bighorn Sheep as well as other alpine-dependant species.

### ***Data Sources***

Information on life history was taken predominantly from primary literature, particularly Wehausen (1980), and also from the summary of bighorn ecology in the 2007 recovery plan. Management history and current practices were summarized from the 2007 recovery plan. Information on population status came from monitoring reports issued by the Sierra Nevada bighorn sheep recovery program led by CDFG and from NatureServe. Information from the quarterly reports of the recovery program were used throughout the manuscript. Spatial data sources include the California Natural Diversity Database (CNDDB 2010) and the Sequoia and Kings Canyon wildlife observation database (SEKI 2010).

## A.7. The Fisher – *Martes pennanti*

### **Abstract**

The fisher (*Martes pennanti*) is a rare carnivore that is primarily associated with mature forest habitats at low to middle elevations (1,000 – 2,000 m) on the western slope of Sequoia-Kings Canyon National Parks. As a result of historic trapping and habitat loss, the range of this species has contracted in California and population estimates for the southern Sierra Nevada are quite low. Although fishers still occur within the parks, individuals require extensive areas of suitable forest; thus, long-term conservation of this species in the parks will only be accomplished in conjunction with efforts to protect this species and its habitat in the region.

- **Species global status:** G5T2T3Q (globally secure with imperiled local populations), Rounded global status for west coast population is T2 (imperiled), Candidate species (12 Sept 2006)
- **Park status:** Rare. Fishers occur at low densities and are restricted primarily to montane hardwood-conifer and mixed conifer forest on the west slope of the parks.
- **Overall integrity:** Low. As a result of protection from logging and preservation of ecological processes (e.g., fire), fisher habitat in the parks may largely resemble historic conditions except where roads and localized development occur on the landscape. However, because this species requires large areas of mature forest, loss and degradation of habitat outside of the parks can negatively affect integrity within the parks. Other concerns in the region include: 1) low genetic diversity, 2) low population estimates, 3) geographic isolation, and 4) overlap in distribution with areas of high fire risk.
- **Certainty of integrity:** Moderate
- **Metrics to evaluate integrity:** Recent surveys have documented distribution within the parks (Green 2007) and the region (Zielinski et al. 2005). Population size has been estimated (Spencer et al. 2007 and 2010) and genetic diversity has been assessed for the region (Drew et al. 2003, Wisely et al. 2004).
- **Vulnerability with respect to key stressors:** Increased risk of mortality due to disease, vehicle collision, and/or predation could impact local populations. Land-use changes and altered fire regimes outside of the parks may pose threats to long-term persistence in the region. Potential effects of climate change on this species and its habitat are complex and the outcomes uncertain.

### **Background and taxonomy**

The fisher (*Martes pennanti*) is a relatively rare carnivore that occurs only in North America and is strongly associated with mature forest habitat (Powell et al. 2003). Fishers are generally a little larger than a house cat with slender weasel-like bodies, triangular heads, rounded ears, and long bushy tails (Powell and Zielinski 1994; Powell et al. 2003). Their fur, highly prized by trappers, ranges in color from chocolate-brown to black with blond highlights on the head and shoulders (Figure A7.1). Fishers are sexually dimorphic, with adult males reaching body sizes over twice that of females. This elusive mesocarnivore is extremely agile in trees, but also travels and forages on the ground. Historical fur trapping and the loss of suitable mature forest habitat have led to



range reductions and local extirpations of this species in some geographic areas. In the United States, the distinct west coast fisher population has undergone a substantial range reduction over the last century and is classified as “warranted but precluded” from listing under the Endangered Species Act (U.S. Dept. of Interior 2004). There is particular concern for the long-term persistence of fishers in the southern Sierra Nevada because they have become geographically and genetically isolated from populations in northern California (U.S. Dept. of Interior 2004; Wisely et al. 2004).

The fisher is in the kingdom *Animalia*, phylum *Craniata*, class *Mammalia*, order *Carnivora*, family *Mustelidae*, and genus *Martes* (NatureServe 2010). The American marten (*Martes americana*) and fisher are the only members of the genus in North America, although other *Martes* species occur in Europe and Asia. The fisher is the largest member of the *Martes* genus and recent genetic evidence suggests it may be more closely related to wolverine (*Gulo gulo*) than other *Martes* species, thus an argument has been made to re-assign it into the sub-genus *Pekania* with wolverine (Stone and Cook 2002; NatureServe 2010). Lastly, fisher subspecies are not generally recognized in North America, however genetically discrete and geographically isolated population segments occur across the range in North America (Drew et al. 2003, NatureServe 2010).

### **Life History**

**Habitat.** Fishers are generally associated with mature forest habitats characterized by large diameter trees, decadent features, and extensive canopy cover (Buskirk and Zielinski 2003). In some areas, they occur on lands managed for timber where remnant structures and adequate canopy cover have been retained. As fishers are habitat specialists with large home ranges for their body size, populations require sizable areas of suitable habitat to survive (Buskirk and Zielinski 2003); a telemetry based study in the southern Sierra Nevada reported mean home range size of 3,934 ha for males and 980 ha for females (Zielinski et al. 2004a). At broad spatial scales, fisher occurrence is associated with extensive areas of mature forest and high percent canopy cover (Carroll et al. 1999; Buskirk and Zielinski 2003), while fine scale resting habitat is characterized by large decadent structures and dense canopy cover (Zielinski et al. 2004b; Yaeger 2005; Purcell et al. 2009). In comparison, little is known about foraging habitat. In California, fishers occur largely in low elevation forests (~900 to 1400 m) composed of hardwoods and conifers as well as mid-elevation forests (~1400 to 2000 m) dominated by conifers (Grinnell et al. 1937; Zielinski et al. 2004a). In the southern Sierra Nevada, forests occupied by fisher contain white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*Pinus jeffreyi*), sugar pine (*Pinus lambertiana*), giant sequoia (*Sequoiadendron giganteum*), and black oak (*Quercus kelloggii*; Green 2007; Purcell et al. 2009).



Figure A7.1. Photograph of a fisher at a remote camera station near Grant Grove, Kings Canyon National Park.



Fishers rest in large live trees, snags, and logs with microsites (e.g., cavities, large limbs, platforms, broken tops, stick nests) that offer some protection from predators and inclement weather (Zielinski et al. 2004b; Purcell et al. 2009). In the spring, females with young localize to den trees with cavities that provide thermal and physical protection for their kits (Powell et al. 2003). Females give birth and initially house their young within a natal den, then move offspring to a series of maternal den structures. The number of structures used by individual females during the den season varies, but may range from 1 to 6 structures. To date, tree species used as dens by females in the southern Sierra Nevada include black oak, white fir, incense cedar, ponderosa pine, and sugar pine (Thompson et al., unpublished data; Sweitzer et al., pers. com.).

*Diet.* Fishers do not hibernate and their lean bodies hold limited fat reserves, so they must locate food on a near daily basis throughout the year. Although largely carnivorous, the fisher diet varies geographically and seasonally (Powell and Zielinski 1994; Zielinski et al. 1999). Porcupines and snowshoe hare are important components of the fisher diet in the northeast, however these large prey are rare or unavailable in the southern Sierra Nevada. Perhaps as a consequence, the fisher diet is quite diverse in this region (Zielinski et al. 1999). Small mammals comprise the bulk of the diet, but medium-sized mammals, birds, bird eggs, lizards, fruits, and carrion are also eaten (Powell and Zielinski 1994; Zielinski et al. 1999). Analysis of scat samples from the southern Sierra Nevada determined that mammals were consumed most frequently, but reptiles, insects, birds, plants (fruits), and fungi were seasonally important (Zielinski et al. 1999).

*Reproduction and Survival.* Females give birth in early spring and mating occurs in the weeks following parturition. After mating, females undergo delayed implantation which inhibits development until mid-winter of the following year (Powell et al. 2003). Thus, females may mate as early as one year of age, but cannot give birth until they are at least two. Number of offspring per female ranges from 1 to 5; preliminary results from the southern Sierra Nevada suggest a mean of 1.5 for this geographic area ( $n = 21$ ; Green et al., unpublished data). On-going research projects in California have identified parturition dates as early as March 23 and den structure use as late as mid- June (Green et al., unpublished data; Sweitzer et al., pers. com., Matthews et al. 2007). Kits stay with their mothers throughout the summer, becoming independent during the fall or winter of their first year (Powell et al. 2003).

Data on survival and dispersal in west coast fisher populations are limited, but preliminary results from projects in the southern Sierra Nevada indicate that as a group young males may have the highest rates of mortality while individual adult females can live to 10+ years (Thompson et al., unpublished data). Age structure data is still being collected, with adult ages ranging from 2 to at least 11 years (Matthews et al., 2007; Thompson et al., unpublished data). Sources of mortality reported to date in California include disease (M. Gabriel, unpublished data), predation by larger carnivores (Wengert et al. 2009), and vehicle collision (Sequoia-Kings Canyon NPS records, Yosemite NPS records, Thompson et al., unpublished data).

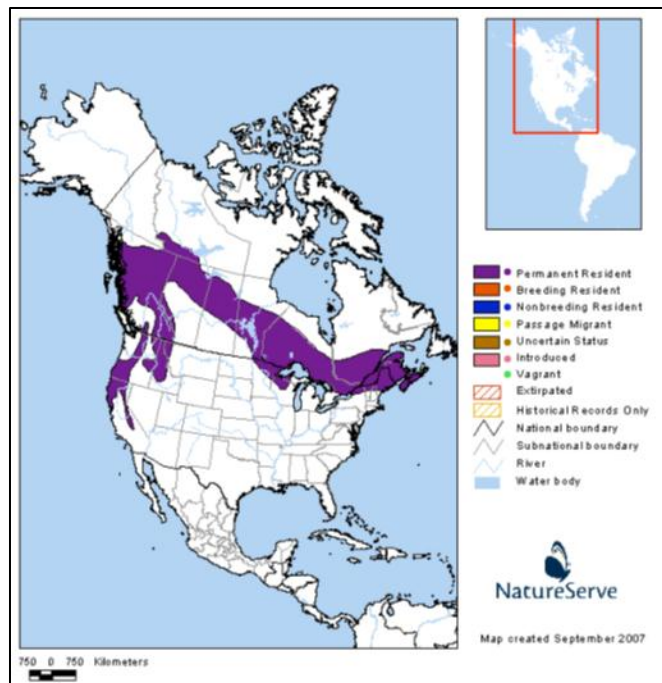
## ***Distribution***

*Global.* The fisher occurs only in North America and currently persists in fragmented populations on the west coast from California north into British Columbia, eastward across the boreal forests of Canada, and in parts of the northeastern United States where suitable habitat remains (Powell 1981; Figure A7.2). Prior to European settlement, fishers occurred more extensively throughout coniferous forests in North America, including parts of Appalachia and Midwestern states southwest of their current distribution (Powell 1981). Trapping and habitat loss have led to fragmentation and local extirpations throughout the range, however, habitat restoration and reintroductions have resulted in sustainable populations in some geographic areas (Powell 1981).

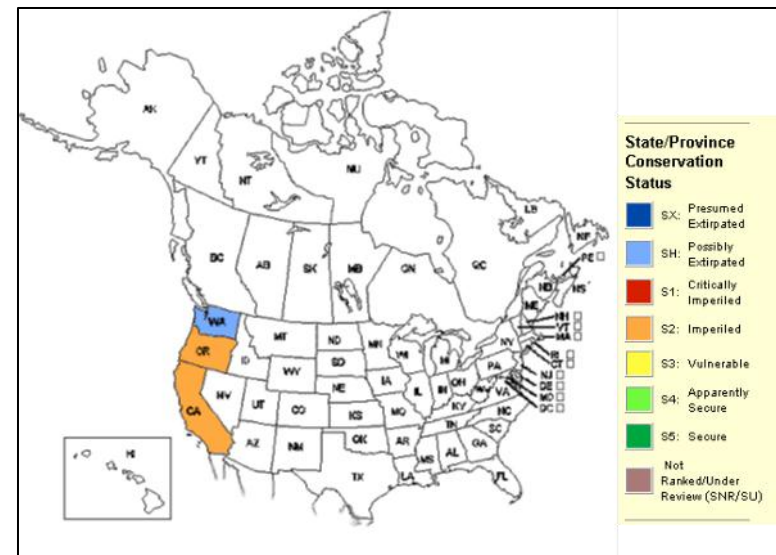
*Regional.* In the western United States, fishers occurred historically in mature forests of Washington, Oregon, and California (Figures A7.2 and A7.3), but their range has been drastically reduced over the last century. Fishers were extirpated from Washington (Aubry and Houston 1994), but were recently reintroduced to Olympic National Park. Two isolated populations exist in Oregon: a native population in the northern Siskiyou Mountains and an introduced population in the southern Cascades (Aubry and Lewis 2003). In California, fishers occur in northwestern coastal forests and the southern Sierra Nevada, but a substantial unoccupied gap now separates the two areas (Zielinski et al. 2005). Emerging genetic evidence is suggesting that this gap may be historic (D. Gammons, pers. Comm.) and may exist for unknown reasons that are independent of human drivers.

Historical records exist for fisher throughout much of the Sierra Nevada, but with a higher concentration in the southern half of the mountain range (Grinnell et al. 1937; Figure A7.4). This pattern of occurrence may simply be associated with variation in habitat suitability across the landscape, but could reflect availability of historical records. Europeans trapped in the northern Sierra for many years before Grinnell and his colleagues compiled their data, potentially biasing distributional records towards areas which had not yet been trapped as intensively (Zielinski et al. 2005). Extensive contemporary track plate and remote camera surveys have demonstrated a substantial reduction in the distribution of fisher in the Sierra, with the current known northern limit near the Merced River in Yosemite National Park (Zielinski et al. 2005; Yosemite NPS records; Figure 4).

*Sequoia and Kings Canyon National Parks.* Historical records for fisher within Sequoia and Kings Canyon National Parks have been summarized to a large extent by Grinnell et al. (1937) and Schempf and White (1974). Because these records include unverifiable observations and trapping records whose locations are often imprecise, interpretation should be viewed with caution. Records from the parks' observation database should also be evaluated prudently, as fishers can easily be mistaken for martens (or martens for fishers) by inexperienced observers, and locations are often generalized. However, the habitats, elevation range, and geographic distribution associated with most records (particularly the historical accounts) are comparable to verified contemporary records for the area (Zielinski et al. 2005; Green 2007; Figures A7.4 and A7.5).



**Figure A7.2.** Current global distribution of the fisher (NatureServe 2010).



**Figure A7.3.** Distribution and conservation status of the west coast fisher population (NatureServe 2010).

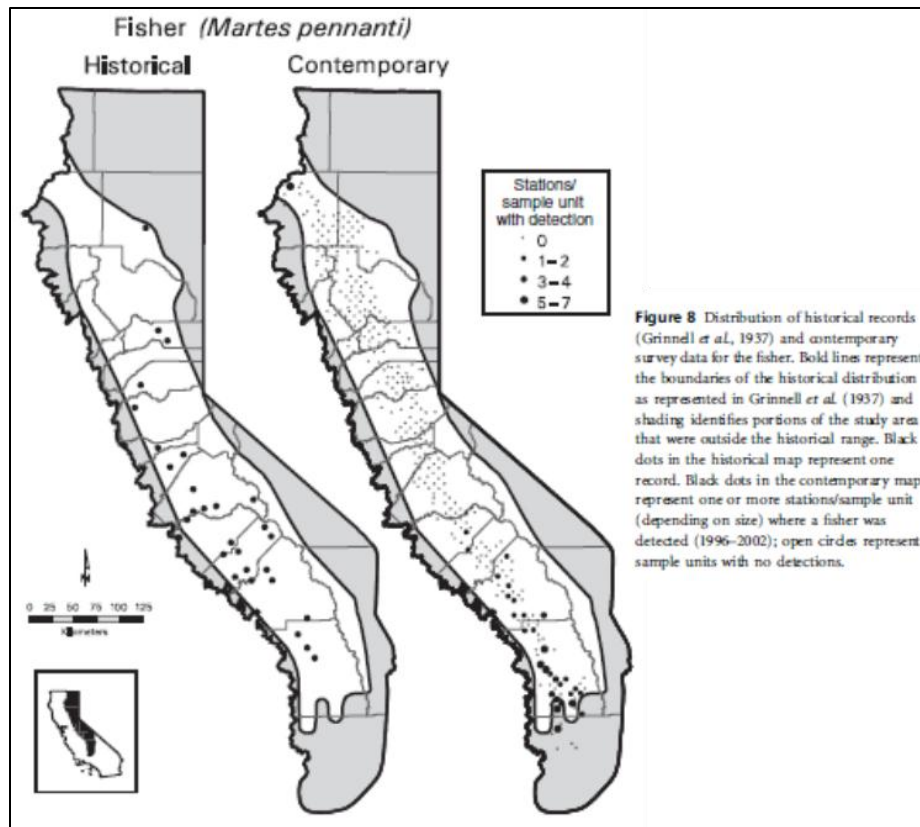
Green (2007) conducted systematic non-invasive surveys using track plate and remote camera stations to document the occurrence of small carnivores across the parks in 2002, 2003, and 2004. Fishers were detected at only 9 of 79 sites surveyed, while American martens were detected at 29 of 79 sites. Fishers occurred in elevations ranging from 990-2870 m (mean 1949, SD = 596). They occurred most frequently in Sierra mixed conifer forest, but were also found in montane hardwood-conifer, white fir, and subalpine forest habitats (based on California Wildlife Habitat Relationship categories). All sites with detections had a minimum of 40% canopy cover. The combined distribution of observations since 1932 and detections from recent surveys suggest that the largely contiguous mid-elevation forest on the western slope is the most consistently occupied, and thus potentially the most suitable, band of habitat for fishers in the parks (Figure A7.5).

### ***Conservation Status***

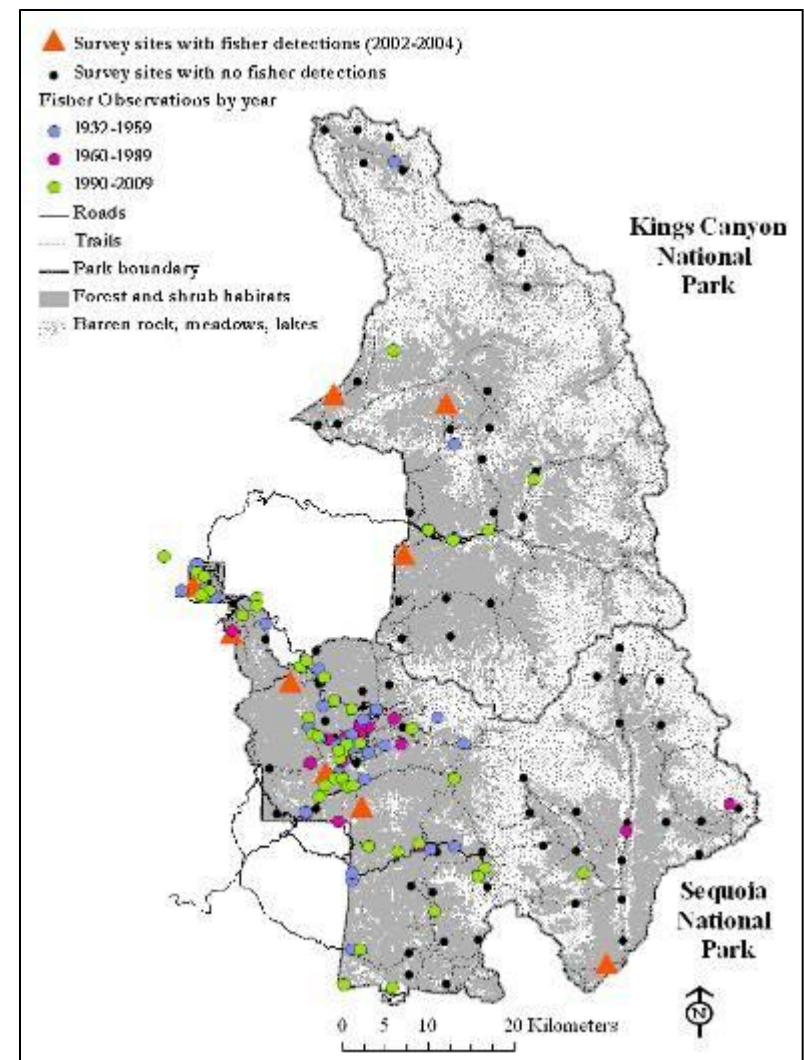
The global conservation status of the distinct west coast fisher population is imperiled (NatureServe 2010). After several petitions, this population segment was designated as “warranted but precluded” from listing as threatened under the federal endangered species act (U.S. Dept. of the Interior 2004). In California, the Department of Fish and Game recently decided not to list the fisher as a threatened species, despite evidence to support listing (McCamman 2010). Region 5 of the United States Forest Service, which includes National Forests in California, classifies the fisher as a species of concern and addresses conservation of habitat in management plans (Powell and Zielinski 1994, North et al. 2009). The population in the Southern Sierra Nevada is of particular conservation concern; using a habitat model combined with contemporary distribution data, Spencer et al. (2007, 2010) estimated that only 220 to 360 fishers remain in the southern Sierra Nevada.

### ***Threats***

The California Department of Fish and Game (2010) identified small population size and risk of severe fire as two of the most significant threats to fishers in the southern Sierra Nevada. Risks associated with roads, predation, disease, poaching, and climate change were considered secondary threats. Historically, the primary threats to fisher conservation were fur trapping and loss of suitable habitat (NatureServe 2010). Although lethal trapping of fishers in western states is now prohibited, habitat loss and alteration continues in many areas. Timber harvest generally does not occur in Sequoia-Kings Canyon, however logging occurs on lands adjacent to the parks thus may influence movements of individual animals and landscape level habitat suitability in the region.



**Figure A7.4.** Comparison of historical and current records of fisher in the Sierra Nevada (figure from Zielinski *et al.* 2004).



**Figure A7.5.** Distribution of fisher observations from database and verified fisher detections from surveys conducted from 2002-2004.

In California, decades of fire suppression have led to an increased risk of large and severe wildfires, which can result in the loss or degradation of suitable fisher habitat (Spencer et al. 2008). Prescribed fire and thinning are often used to reduce the risk of catastrophic wildfire. While mechanical thinning and controlled burns can indirectly benefit fishers by promoting forest resilience and reducing risk of catastrophic wildfire, there may be direct negative effects to fishers locally (Spencer et al. 2008). Seasonality of treatments may result in differential effects on fishers; if these activities occur in the spring when kits are vulnerable they could result in kit mortality. On the other hand, varying the seasonality of management activities has the potential to benefit fishers by promoting prey diversity (Knapp et al. 2009).

Mortalities associated with roads have been documented in Sequoia-Kings Canyon and Yosemite National Parks. Between 2002 and 2005 at least three car-related fisher mortalities occurred in the western half of Sequoia-Kings Canyon: one in Giant Forest (R. Green, pers. obs.) and two in Grant Grove (R. Mazur, pers. com.). In Yosemite, at least seven road-related mortalities were reported between 1993 and 2010 in the southeastern portion of the parks – including two lactating females (S. Stock and L. Chow, pers. com.). So even though park lands are protected, busy roads cutting across fisher habitat can result in road mortalities that impact local populations. Other activities with potential impacts to individuals and populations include predation, illegal marijuana cultivation (e.g., pesticide use, trapping), and disease (e.g., canine distemper, parvovirus). Geographic isolation and small population size could amplify these and other threats.

### ***Data Gaps and Research Needs in Sequoia-Kings Canyon***

#### Critical Data Gaps and Associated Research Needs

- Comparison of fisher populations inside SEKI vs surrounding landscapes as SEKI manages land without timber harvest and different fire management policies than the surrounding public lands. These may have impacts of fisher population performance.
- Estimates of population size and viability in the parks and surrounding area
- Survival estimates by sex and age class
- Causes of mortality and degree of impact
- Barriers to dispersal
- Description of reproductive ecology and habitat (e.g., reproductive den structures, fecundity, dates kits may be vulnerable to disturbance)
- Description of foraging ecology and habitat – especially in relation to location and availability of rest and den structures
- Investigation into the relationship between fire, forests, and fisher ecology

#### Research Needs Associated with Climate Change

- Assessment of the potential effects of predicted climate related changes (e.g., temperature increase, fluctuation in precipitation patterns, shifts in plant phenology) on fisher distribution, habitat, and long-term persistence
- Modeling potential effects of predicted shifts in distribution of vegetation on fishers in particular, and carnivore and/or prey communities in general

- Evaluation of how changes in precipitation patterns (e.g., snow vs. rain) may alter resource availability and seasonal movement patterns

### ***Summary and Recommendations***

The fisher population in the southern Sierra Nevada has become geographically isolated and potentially vulnerable to extinction due to its low numbers and unique habitat requirements. Although trapping is now illegal and timber harvest generally undergoes stricter review than it once did, these practices have influenced the current distribution of fishers and the spatial arrangement of the remaining suitable habitat in the Sierra Nevada. Much of the forested habitat available to fishers within the parks has been protected for over a century and is currently managed to promote natural ecological processes. However, even within the parks some risks cannot be easily controlled (e.g., accidental fire, influence of climate change, spread of disease) and factors outside park boundaries can alter habitat suitability of the whole region (e.g., timber harvest, habitat fragmentation, risk of catastrophic fire). Finally, although the impacts of some threats are still poorly understood (e.g., disease, predation, road mortality), on-going research will hopefully provide insight into the influence of these factors in the near future.

Because fishers have large home ranges, are relatively rare on the landscape, and require large areas of continuous suitable forest, it seems unlikely that a viable population could persist independently on park lands. Additionally, portions of the parks are largely unsuitable for fishers, as almost the entire eastern half is dominated by high elevation habitats with little or no forest cover. Although Sequoia and Kings Canyon do contain large tracts of protected mid-elevation mature forest, an extensive network of suitable habitat extending onto lands adjacent to the parks is likely needed to promote the long-term persistence of fishers within the parks and surrounding area. Also, as fishers occur in habitat types and elevation zones which continue to be altered by human activity in the region (e.g., timber harvest, development, fire suppression, fire risk reduction), the parks should not assume that fisher occurrence and availability of suitable habitat is invariable on neighboring lands.

In order to promote fisher conservation in Sequoia-Kings Canyon and the region, we suggest the parks consider a three part strategy for this species. First, we recommend that park personnel become and/or stay involved with regional and statewide fisher conservation efforts to stay informed about ongoing research and determine how the parks can contribute to conservation plans. A recently formed southern Sierra Nevada fisher working group, composed of individuals from a variety of organizations, would be a good place to start (contacts: Kathryn Purcell and Greg Schrorer). Second, we suggest that periodic monitoring of fisher occurrence in the parks be conducted – either by park personnel or in conjunction with other long-term monitoring projects. These efforts could target fisher specifically or be designed to detect a number of carnivore species of interest (e.g., American marten, wolverine, Sierra Nevada red fox). Previously used monitoring strategies (e.g., Green 2007) could be repeated or altered to incorporate higher quality remote cameras and/or hair snares. Surveys could also be re-designed to better assess the potential long-term effects of climatic change. Finally, we recognize that the parks provide a unique environment in which to study the relationship between fishers and fire (prescribed and naturally ignited) as well as habitat use in a relatively pristine old growth forest ecosystem; research projects addressing these topics could be conducted by park personnel and/or collaborative researchers. While the parks may not have the resources to implement all of the

steps proposed here, much of this work could be combined with long-term monitoring of other species, collaboration with researchers, and participation by various park personnel (e.g., backcountry rangers).

### **Data Sources**

Species life history information was gleaned from a variety of sources including NatureServe Explorer (*Martes pennanti* and *M. pennanti* population 1), United States Fish and Wildlife Service documents, California Fish and Game documents, United States Forest Service technical reports, the Sequoia-Kings Canyon Wildlife Observation database, and the primary literature. National Park Service employees from Sequoia-Kings Canyon and Yosemite provided information on specific records (e.g., road kills). Unpublished information from on-going research projects in California was also included and associated publications should be available in the near future. Those sources include:

- Kings River Fisher Project, Pacific Southwest Research Station, United States Forest Service: Dr. Craig Thompson, Dr. Kathryn Purcell, Rebecca Green (UC Davis graduate student), and James Garner (Humboldt State graduate student)
- SNAMP (Sierra Nevada Adaptive Management Program) Fisher Project, UC Berkeley: Dr. Rick Sweitzer and Dr. Reginald Barrett
- Hoopa Fisher Project, Hoopa Tribal Forestry: Mark Higley (Hoopa Tribal Forestry) and Sean Matthews (the Wildlife Conservation Society)
- Greta Wengart (UC Davis graduate student): using forensic techniques to identify predators of fishers on several projects in California
- Mourad Gabriel (UC Davis graduate students: assessing prevalence of disease in fisher populations across California



## A.8. Mountain Yellow-legged Frogs: *Rana muscosa* and *Rana sierrae*

### **Abstract**

Mountain yellow-legged frogs are endemic to the Sierra Nevada and Transverse Ranges of southern California, and presently have critically low populations within SEKI. Populations are currently threatened by a variety of concerns including introduced fish and disease. Although once lumped together as a single species, mountain yellow-legged frogs have been split into *Rana muscosa* (aka the southern mountain yellow-legged frog) and *R. sierrae* (aka the Sierra Nevada yellow-legged frog). Both species occur within Sequoia and Kings Canyon National Parks and have disappeared from most of their native habitat.

Scientific observations of mountain yellow-legged frog populations within the Sierra Nevada circa 1915 were well documented by Camp (1917) and Grinnell and Storer (1924), although museum specimens exist from as early as 1899 (Vredenburg et al. 2007). In particular, the last 20 years of monitoring by the NPS, USGS, California Department of Fish and Game, and University of California has yielded fine scale detail of population fluxes. From 1997-2002, 558 water bodies (lakes and ponds), out of approximately 3200 water bodies surveyed in Sequoia and Kings Canyon National Parks contained mountain yellow-legged frogs. By 2007, resurveys of the 558 occupied water bodies showed that the number of occupied sites had declined by 46% to 254 water bodies (R. Knapp, 2008 data report).

Although the southern California population of *R. muscosa* is federally listed as endangered, the Sierra Nevada populations of both species are federal candidate species. Before the species split, the USFWS listed the DPS (distinct population segment) of (former) *R. muscosa* south of the Tehachapi mountains as endangered, but the frogs north of the Tehachapi were “warranted but precluded” by “higher priority actions”. The Sierra Nevada population DPS (*R. muscosa* + *R. sierrae*) was given a priority listing of 3 by the USFWS, stating that “we must spend nearly all of our Listing Program funding to comply with court orders and judicially approved settlement agreements, which are now our highest priority actions” (Federal Register, 16 January 2003). In the mean time, the Sierra Nevada DPS was added to the candidate species list.

- **Species global status:** *Rana muscosa*: G2. Imperiled (G2)— At high risk of extinction; US Listed endangered. *Rana sierrae*: G1G2. Critically Imperiled (G1)—At very high risk of extinction. US ESA candidate taxa.
- **Park status:** Rare, at risk of extinction
- **Overall integrity:** Very Poor.
- **Certainty on integrity:** High.
- **Metric to evaluate integrity:** Abundance, in terms of presence and absence of populations within breeding habitat (lakes) as well as population size at individual lakes.
- **Vulnerability with respect to key stressors (air quality, land use change, climate change, invasive species, altered fire regimes, disease):** *R. muscosa* and *R. sierrae* populations are extremely sensitive to introduced trout and the fungal pathogen chytridiomycosis. Contaminant introductions, acidification, livestock grazing, UV radiation, drought, recreation, timber harvest, water diversions, and fire management activities may also contribute to population declines (USFWS 2003), as well as climate change (Lacan et al. 2008). However, acidification (Bradford et al. 1994a) and UV

radiation (Vredenburg 2002) have been shown to not be significant threats.

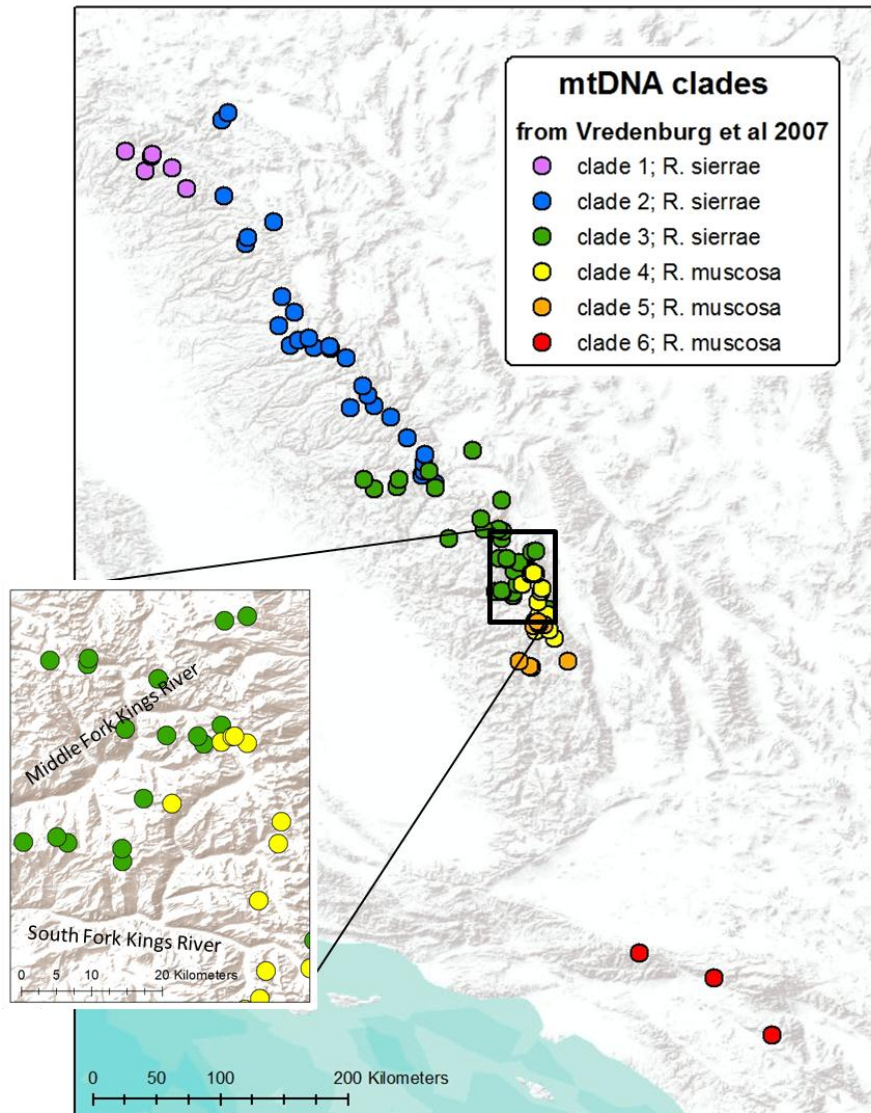
### **Species Background/Life History**

The mountain yellow-legged frog is in the family: *Ranidae*, subgenus: *Amerana*, genus: *Rana*. However, it has been reclassified several times at the species level. The type specimens were originally described by Camp as two separate subspecies of *Rana boylei*: *R. boylei muscosa* and *R. boylei sierrae* (Camp 1917). These morphological classifications remained until 1955 when the subspecies were grouped together and raised to species status with the name *R. muscosa* (Zweifel 1955). Due to recent declines, in 2002 USFWS listed the few disjunct populations remaining in the transverse ranges of southern California as an endangered “distinct population segment” (DPS) of *Rana muscosa* under the Endangered Species Act (USFWS 2002). At this time, the species *R. muscosa* included contained all populations of mountain yellow-legged frog, from the northern Sierra Nevada to the mountains around Los Angeles basin. The most recent taxonomic work, by Vredenburg and others (2007), combined genetics, morphology, and vocalizations to identify two major clades, each encompassing three minor clades (Figure 1). This work supported the division of the mountain yellow-legged frog into two distinct species within the Sierra Nevada range, and named more southerly populations *R. muscosa* (containing populations both in southern California and the southern Sierra Nevada) and the more northerly *R. sierrae* in the Sierra Nevada only. The contact zone of these two new species lies in SEKI between the Middle and South Forks of the Kings River (Vredenburg et al 2007). For the remainder of this report, *Rana muscosa* and *Rana sierrae* will still be discussed using the same common name, mountain yellow-legged frogs.

Both of these species have similar life-histories, with slight variations in the populations of the transverse ranges. Adult frogs are primarily insectivorous and aquatic, long believed to remain within 1m of water (Mullally and Cunningham 1956, Bradford et al 1993). In the past decade, however, they have been observed to move nearly a kilometer overland (Matthews and Pope 1999, Pope 1999, Vredenburg 2002) and terrestrial movements may be more common than previously thought. The home range of mountain yellow-legged frogs varies widely over a single season and among individuals in Sequoia and Kings Canyon National Parks. In August, home ranges for ten females varied from 19.4–1,028 m<sup>2</sup> ( $x = 385.5$  m<sup>2</sup>; s.e. = 113.4 m<sup>2</sup>) using adaptive kernel 90% contours. In September, ranges for seven females varied from 53–9,807 m<sup>2</sup> ( $x = 5,099$  m<sup>2</sup>; s.e. = 1,506 m<sup>2</sup>), and one female was calculated at 6,990 m<sup>2</sup> (Matthews and Pope, 1999). In the high elevations of the Sierra Nevada, mountain yellow-legged frogs overwinter in ice-covered lakes and streams that are deep enough to prevent oxygen depletion (Bradford 1983, Bradford 1989), although there is evidence for adult winter-survival in ponds that freeze to the bottom (Matthews and Pope 1999, Pope 1999). Soon after the ice has melted from these water bodies, usually between April and July, the adults emerge to begin breeding and some may make short migrations to better breeding habitats (Matthews and Pope 1999, Pope and Matthews 2001). After the breeding season, habitat selection by adult frogs is likely motivated by feeding needs; more were found in warmer, shallow lakes (<1.5m) in the month of August than in late September as frogs moved to deeper lakes to overwinter (Matthews and Pope 1999, Finlay and Vredenburg 2007).

Mountain yellow-legged frog larvae will remain pre-metamorphic for two or more summer seasons after they hatch, dependent upon the elevation (Storer 1925, Zweifel 1955, Bradford

1983). Because of this extended larval stage, tadpoles of the mountain yellow-legged frog require lakes that are permanent throughout the summer. The larvae are also more likely to survive to metamorphosis in lakes which retain moderate levels of dissolved oxygen in winter months (Bradford 1989), although larvae have a higher tolerance of low oxygen conditions than the adults, and have been reported to survive even when shallow lakes freeze to the bottom for months (Bradford 1983, Matthews and Pope 1999).



**Figure A8.1:** Map adapted from Vredenburg et al 2007. Points represent localities of genetic samples of mountain yellow-legged frogs. Seven individuals from the Museum of Vertebrate Zoology tissue bank (UC Berkeley), 8 from California Academy of Sciences, and 76 from live individuals captured in the field. Colors correspond to the six clades identified; clades 1-3 belong to *Rana sierrae*; clades 4-6 belong to *R. muscosa*.

### **Global and Regional Context/Population Genetics**

The mountain yellow-legged frog complex is endemic to the west coast of North America, with a regional distribution that at one point included populations in the northern Sierra Nevada of

California (Butte Co.), southern California in the San Gabriel, San Bernardino, and San Jacinto Mountains, and east just into Nevada in the area of Lake Tahoe (Figure A8.1; Zweifel 1955). Its elevation range extends from 1370 m to 3650 m on the west slope of the Sierra Nevada, and from 370 m to 2290 m in the southern California populations (Zweifel 1955, Mullally and Cunningham 1956).

*Rana muscosa* and *R. sierrae* have experienced widespread declines over the past century. Once described as the most common amphibian in regions of the Sierra Nevada (Grinnell and Storer 1924), their decline has been noted by many researchers (Bradford 1994b, Jennings and Hayes 1994, Drost and Fellers 1996, Knapp 1996, Knapp and Matthews 2000) with one estimate of a 94% extinction rate based on extensive and widespread surveys of historic collection localities (Vredenburg et al 2007). Biodiversity conservation groups such as Conservation International and NatureServe have classified this species complex as critically threatened (IUCN: Endangered; NatureServe: G2 – Imperiled).

This dramatic decline over the past century has occurred despite the presence of most of the range of the mountain yellow-legged frog on public, protected lands (Knapp and Matthews 2000). It appears that the populations on NPS lands (Yosemite, Sequoia, and Kings Canyon National Parks) have experienced a more moderate decline than others (USFWS 2003, Knapp 1996), likely due to historically lower intensity of fish-stocking (Knapp and Matthews 2000). Even so, Bradford et al. (1994b p. 325) estimate that by the late 1980s roughly half of the historic populations in SEKI had been extirpated. By 2002, only 558 lakes of approximately 3200 surveyed in the entire park contained frogs, and by 2007 that number had declined by 46% to 254 lakes (Knapp 2008)

In addition to containing several basins that support mountain yellow-legged frog populations, SEKI encompasses the contact zone of *R. sierrae* and *R. muscosa*, indicating a large amount of genetic variation represented within the park (Vredenburg et al 2007). Between this relatively larger population and increased genetic diversity, SEKI has a unique opportunity to contribute to the persistence of the mountain yellow-legged frog.

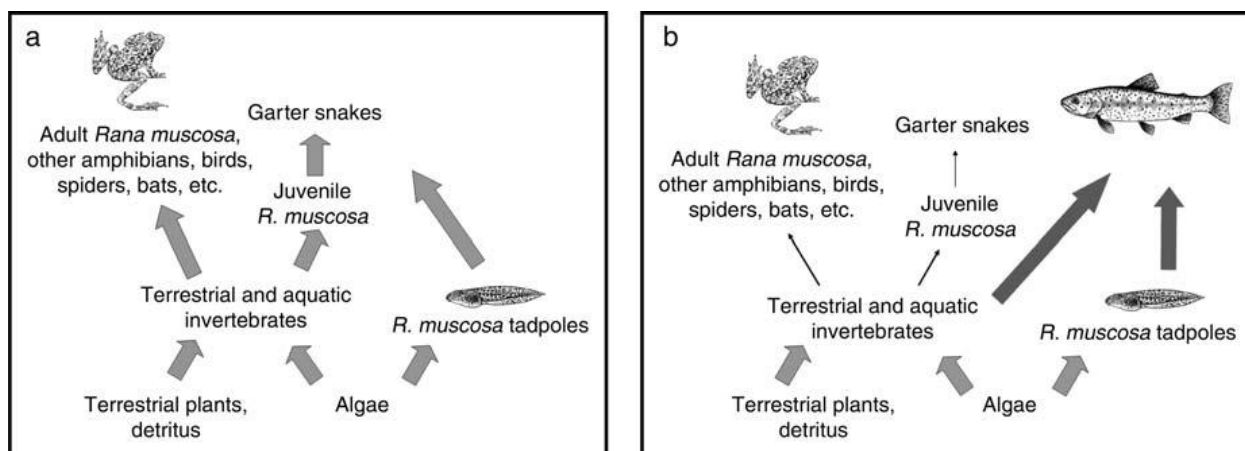
### **Population Threats**

The major factors implicated in the decline of the mountain yellow-legged frog include the introduction of trout into naturally fishless lakes, the deposition of airborne chemicals from the Central Valley, and the recent epidemic of chytrid fungus. Additional threats have been identified, including livestock grazing, UV radiation, drought, recreation, timber harvest, water diversions, and fire management activities (USFWS 2003). These threats exist throughout the range of these species.

*Introduced Trout.* Historically trout were excluded from most water bodies in the Sierra Nevada above 1800m, due to impassible barriers (Knapp 1996, Moyle 2002). However, beginning in the mid 1800s humans began to stock various trout species into these high elevation lakes to create recreational fisheries (Knapp 1996, Pister 2001). Stocking began on foot or by pack animal, and included primarily brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), and various strains of rainbow trout (*Oncorhynchus mykiss*) (Balhs 1992, Pister 2001). Beginning around 1950, CDFG became the primary agency responsible for planting fish, and began to utilize low-

flying planes to access remote areas (Pister 2001). CDFG continues to stock trout on most public lands with the purpose of maintaining existing populations, however stocking was terminated in Sequoia and Kings Canyon National Parks in 1988 (Knapp 1996). Throughout the Sierra Nevada range it is now estimated that 63% of all water bodies contain introduced trout (Knapp 1996), while roughly 45% of naturally fishless lakes in SEKI contain introduced trout (Bradford et al 1993). The proportion is lower for SEKI because NPS adopted a policy against active stocking in 1972 (NPS 1975). While a majority of stocked populations were self sustaining, some were not (Zardus 1977, Armstrong and Knapp 2004), resulting in a natural decline of lakes inhabited by introduced trout.

The presence of introduced trout has posed a large threat to the mountain yellow-legged frog. Both trout and mountain yellow-legged frog require permanent water bodies that provide adequate oxygen throughout the winter months when the lakes are frozen over. Thus mountain yellow-legged frog tadpoles and fish are both found in lakes >1ha surface area and >1.5m deep. However, trout and frogs rarely exist in the same lakes or streams (Grinnell and Storer 1924, Bradford 1989, Bradford et al 1993, Drost and Fellers 1994). Fish were at first assumed to exclude frogs through predation of tadpoles and young adults (Grinnell and Storer 1924, Bradford 1989). Fish predation on mountain yellow-legged frog was then experimentally confirmed by Vredenburg (2004); and frog population suppression by fish was measured by Knapp and others (2007), who showed that populations significantly expanded immediately following fish removal. Because the larvae require permanent water bodies and at least two summer seasons before metamorphosis (Zweifel 1955), they experience a long period of predation pressure if coexisting with fish. In addition to directly consuming tadpoles, trout alter the aquatic insect and zooplankton communities within historically fishless lakes, and thus may reduce or eliminate the invertebrates important in the diet of mountain yellow-legged frog adults (Figure A8.2, from Finlay and Vredenburg 2007).



**Figure 8.2:** Foodweb diagram contrasting energy flow pathways to consumers under (a) historical, fishless conditions and (b) as influenced by introduced trout. The size of the arrows indicates the relative strength of trophic connections between food web components. Darker arrows indicate important changes in the flow of energy to fish in lakes where trout have been introduced compared to fishless lakes. From Finlay and Vredenburg (2007).

In addition to direct mortality and resource competition, non-native trout fragment populations of mountain yellow-legged frog. Bradford et al. (1993) estimated that the introduction of trout decreased the connectivity between populations by an estimated ten-fold. However, this estimate may be slightly high because it is assumed that frogs could only migrate between sites through aquatic pathways, while more recent work on the movement of adults has shown a higher ability for terrestrial movement than previously believed (Matthews and Pope 1999). Even if the estimate is slightly high, the increased fragmentation poses an additional management dilemma for both NPS and other federal agencies working to restore populations of mountain yellow-legged frog (Bradford et al 1993). Like many temperate amphibians, the mountain yellow-legged frog can experience large fluctuations in local population size (Pechmann et al 1991). When the individual populations can act as one metapopulation (Hanski and Gilpin 1991), the species persists through the replacement of extirpated populations, or the rescue of small populations (Bradford et al 1993, Drost and Fellers 1996, Knapp and Matthews 2000). With increased fragmentation reducing immigration, the persistence of small populations will be decreased, along with the likelihood of the natural re-establishment of extinct populations (Hanski and Gilpin 1991).

While it is clear that introduced trout exclude frogs from much of their historic habitat, recent declines may not be able to be explained by fish stocking alone. However, the reduction in population size and connectivity caused by fish likely make the mountain yellow-legged frog less resilient to additional threats such as disease and contaminants, as discussed below.

*Disease.* A lethal disease, chytridiomycosis, caused by the aquatic fungal pathogen *Batrachochytrium dendrobatidis* (Berger et al. 1998) is responsible for recent population extinctions in *R. muscosa* and *R. sierrae* in the Sierra Nevada (Rachowicz et al. 2006). Transmission of the disease can occur within and among life stages in *R. muscosa* and *R. sierrae* but mortality only occurs for post-metamorphic adults and juveniles (Rachowicz & Vredenburg 2004). Maximum growth rate of the fungus occurs at 17-25 °C but individuals maintained at 22 °C exhibited lower mortality than those housed at 17 °C (Andre et al. 2008).

Recent studies have shown a limited ability of innate immunoresponses to defend *R. muscosa* and *R. sierrae* against chytridiomycosis. However, populations of *R. muscosa* and *R. sierrae* do vary in their susceptibility to the disease, and the variation may be explained by differences in anti-*B. dendrobatidis* skin bacteria. Anti-microbial skin peptides have been shown to be effective at killing *B. dendrobatidis* in *R. muscosa* and *R. sierrae* (Rollins-Smith et al. 2006), however no differences in anti-microbial skin peptides were observed in the field. Due to the frog's inability to resist and survive infection, immunization with a killed chytrid adjuvant mixture has been attempted within an experimental context. Unfortunately, immunization was found to be an ineffective means of preventing infection or reducing mortality in *R. muscosa* and *R. sierrae*.

*Chemical Contaminants.* The role of chemical contaminants in the population declines of mountain yellow-legged frogs remains unclear. As early as 1997, pesticide concentrations were implicated in the population declines of mountain yellow-legged frogs. Water and tissue samples taken from a failed experimental reintroduction of frogs to the Tablelands were compared with those taken from a healthy, reference population of frogs in Sixty Lake Basin, both locations within Sequoia and Kings Canyon National Parks. The concentration of  $\gamma$ -chlordane and trans-

nonachlor in frog tissues and organophosphate insecticides, chlorpyrifos, and diazinon in surface water samples were higher in concentration at the Tablelands site than the Sixty Lake Basin site (Fellers et al. 2004), which might explain both the original population crash in the Tablelands as well as the failed experimental reintroduction.

More recent assays (2004 and 2005) for contaminants in air, sediment, and tissue of Pacific treefrog (*Hyla regilla*) tadpoles showed that while chemical concentrations detected in sediment or tadpole tissue were very low (averaging in the parts per billion or less in sediment and tadpoles), up to 15 chemicals were detected frequently in tadpoles including current- and historic-use pesticides, polychlorinated biphenols, and polycyclic aromatic hydrocarbons. Only  $\beta$ -endosulfan was identified in the air, at concentrations of approximately 10 pg/m<sup>3</sup>. Samples taken from sites across Sequoia and Kings Canyon National Parks exhibit the same chemical composition and chemical concentrations do not change as a function of distance from San Joaquin Valley (location of pesticide inputs) (Bradford et al. 2010). Based on the low concentrations of chemicals that have been documented within Sequoia and Kings Canyon National Parks, it is likely that if chemical contaminants are playing a role in the population declines of the mountain yellow-legged frog it is probably due to sublethal effects and possible synergistic effects that make the frogs more susceptible to disease or predation (Taylor et al 1999, Gilbertson et al. 2003). It seems that chytridiomycosis and trout predation is capable of causing population extinctions regardless of contaminant levels (Roland Knapp, personal communication).

*Additional Threats.* Population and habitat fragmentation (due to introduced trout and climate change) may also contribute to declines of the mountain yellow-legged frog. The mountain yellow-legged frog displays high site fidelity. During periods of overwintering, breeding, and feeding the probability of returning to previously used water bodies is typically greater than 80% and always greater than the probability of moving to other water bodies (Matthews & Priesler 2010).

General circulation models (GCMs) predict an increase in air temperatures across the Sierra Nevada on the order of 2-6 °C (Dettinger 2005, Hayhoe et al. 2004). While predictions vary as to whether the region can expect more or less precipitation, it is generally accepted that there will be decreases in mean annual flow, reduced snowpack, and more rapid snowmelt as a result of climate change (Medellin-Azuara et al. 2008, Vicuna et al. 2008, Miller et al. 2003, Knowles & Cayan 2002). Certainly, both *R. muscosa* and *R. sierrae* will experience localized changes to their habitat, perhaps rendering it unsuitable for frog survival. The tendency for philopatry indicates that the mountain yellow-legged frog may not have the capacity for unaided migration to suitable habitat, as necessitated by future climate change.

### **Past and Present Management**

In response to population declines of the mountain yellow-legged frog caused by introduced trout predation, Sequoia and Kings Canyon National Parks initiated a restoration management program that sought to eradicate non-native trout from high-priority, high elevations lakes and associated streams within the park boundary in 2001. Prior to implementation of the project, the project proposal and environmental assessment entitled “Preliminary Restoration of Mountain Yellow-legged Frogs” received 80% approval, based on comments submitted to the park superintendent. During the first year of the project six lakes were targeted, including three in

Sixty Lake Basin and three in Upper LeConte Canyon. Five additional lakes were added to the project in 2004 or 2005, including two in Sixty Lake Basin and three in Upper Bubbs Creek. Seven additional lakes were begun in 2009, including two in Sixty Lake Basin, two in Kern Point Basin, two in Upper Basin, and one in Pinchot Basin. Since 2001, nearly 35,000 trout have been removed, eight lakes being completely eradicated and three lakes nearly completely eradicated. In nine of the eleven lakes that remained free from infection by *B. dendrobatidis* three years after trout removal, average population density (tadpoles and frogs per 10 meters of shoreline) increased 14-fold, one lake showed a 49-fold increase (Boiano and Meyer 2010)

One of the greatest challenges to fish eradication in lakes has been the removal of fish from adjoining stream habitat. As such, chemical techniques are being considered to aid in future fish removal. While recognized as an important fishery management tool (McClay 2005), the use of rotenone is controversial. Fish display a relatively uniform response to rotenone application (LD50 after 6 hrs is 3 to 42 lg/L rotenone) (Marking & Bills 1976), but the response by aquatic invertebrates varies widely (LD50 after 6 hrs is 1.8 to 1,700 lg/L rotenone) (Chandler & Marking 1982). As a result, high concentrations of rotenone (>100 µm) and long treatment applications (>8 h) can have very severe impacts on invertebrate communities (Binns 1967, Mangum and Madrigal 1999; Darby et al. 2004) sometimes resulting in complete, short-term annihilation of aquatic macroinvertebrates. However, at lower concentrations (<50 lg/L) and shorter treatment durations (<8 h) rotenone treatments can have less impact on invertebrate assemblages (Cook & Moore 1969; Maslin et al. 1988; Trumbo et al. 2000a, 2000b; Whelan 2002) while still being potentially effective at eradicating fish. Reviewers of rotenone usage have made the following recommendations to land managers in order to maximize effective fish removal while minimizing biodiversity loss among aquatic macroinvertebrate populations.

- “ (1) apply rotenone at treatment rates between 25 and 50 lg/L,  
(2) operate rotenone drip stations for 4–8 h per treatment,  
(3) use unsynergized formulations because the synergized formulation is less toxic to fish and more toxic to aquatic insects,  
(4) for chemical treatments of larger drainages stage treatments with intermediate barriers and allow time between treatments for dispersal and recolonization of invertebrates to avoid potential for cumulative impacts,  
(5) leave headwater reaches of drainages that are above barriers and have never inhabited fish as untreated refuges for invertebrates and a source for recolonization of downstream treated reaches,  
(6) neutralize rotenone downstream of the project area,  
(7) consider aquatic invertebrate “rescues” to probably be impractical except where treating whole or isolated basins or the presence of endangered invertebrate species, and  
(8) strategically use caged sentinel fish and collect water samples for rotenone content throughout the treatment area to monitor efficacy. ” (Finlayson et al. 2010)

Roland Knapp has suggested selecting one lake basin to apply rotenone and then to intensively monitor that basin before, during, and after treatment. Data from this study could then be used to determine whether rotenone could be used in subsequent restoration efforts (personal communication).



**Future Action**

Based on the current threats to populations of *R. muscosa* and *R. sierrae* and the success of restoration efforts, Roland Knapp has suggested that the highest management priority should be to remove as many fish populations as possible from entire lake basins (lakes and associated streams). Persistence of frog populations in the face of disease, climate change, and chemical contaminants will be facilitated by having robust frog populations distributed across as much of their historical habitat as possible. Also, increased emphasis should be placed on reintroducing frogs into previously occupied areas from which they have been extirpated, including much of western Sequoia and Kings Canyon National Parks, including stream and meadow environments. The success of such efforts will be low due to ongoing effects of *B. dendrobatidis*, but reintroductions designed as experiments are essential for increasing our understanding of management options in the face of *B. dendrobatidis* (Roland Knapp, personal communication).

## B. Park Limited Taxa.

### B1. Slender Salamanders: *Batrachoseps kawia* and *Batrachoseps regius*

#### **Abstract**

The slender salamanders represent a difficult group. This is a suite of little known taxa that appear to be isolated into a complex of closely related endemic taxa. Population data of *Batrachoseps kawia* and *Batrachoseps regius* (in terms of species range and population size) are patchy at best, in terms of both temporal and spatial scales. In addition, identification of these two species by morphological characters is very difficult, making the accuracy of the existing population data questionable. We can infer that *B. kawia* and *B. regius* may not be sensitive to climate change as their phylogeography indicates that both species have survived historic climate oscillations of the scale of current climate change projections. There is some evidence to suggest that they may be sensitive to the disease chytridiomycosis, but habitat alteration may pose the most significant threat to *B. kawia* and *B. regius*.

- **Species global status:** *Batrachoseps kawia*: G1G2. *Batrachoseps regius*: G1. Imperiled to critically imperiled.
- **Park status:** Rare
- **Overall integrity:** Marginal
- **Certainty on integrity:** Very uncertain.
- **Metric to evaluate integrity:** Changes in population size and distribution over time.
- **Vulnerability with respect to key stressors (air quality, land use change, climate change, invasive species, altered fire regimes, disease):** Vulnerability of *Batrachoseps kawia* and *Batrachoseps regius* to key stressors is uncertain due to limited information on this subject; possibly vulnerable to disease.

#### **Species Background/Life History**

In California, slender salamanders (*Batrachoseps*, Family: *Plethodontidae*, Subfamily: *Hemidactyliinae*) are primarily terrestrial species that seek moist, protected places during the day, and emerge at night under favorable environmental conditions (high humidity) (Stebbins 2003). *Batrachoseps* salamanders undergo direct development, so young are hatched fully formed, as compared to young that emerge as larvae in other salamander taxa. Eggs are usually deposited underneath logs or leaf litter in the winter (Stebbins 2003). . In general, the movement of plethodontid salamanders is poorly documented, but it is clear that home ranges tend to be very small, on the order of a few meters to a few dozen meters in diameter (Marvin 2001). For example, Welsh and Lind (1995) found that over six months, 80% of *Plethodon elongatus* females and 66% of males were recaptured in the same 7.5 x 7.5 m grid. *Batrachoseps* identification to species is cryptic at best (Stebbins 2003) and accurate identification to species is based almost exclusively on genetic analysis (David Wake, personal communication). Slender salamanders may be particularly vulnerable to chemical pollution due to their permeable skin (Blaustein and Wake 1990; Alford and Richards 1999).

### **Global and Regional Context**

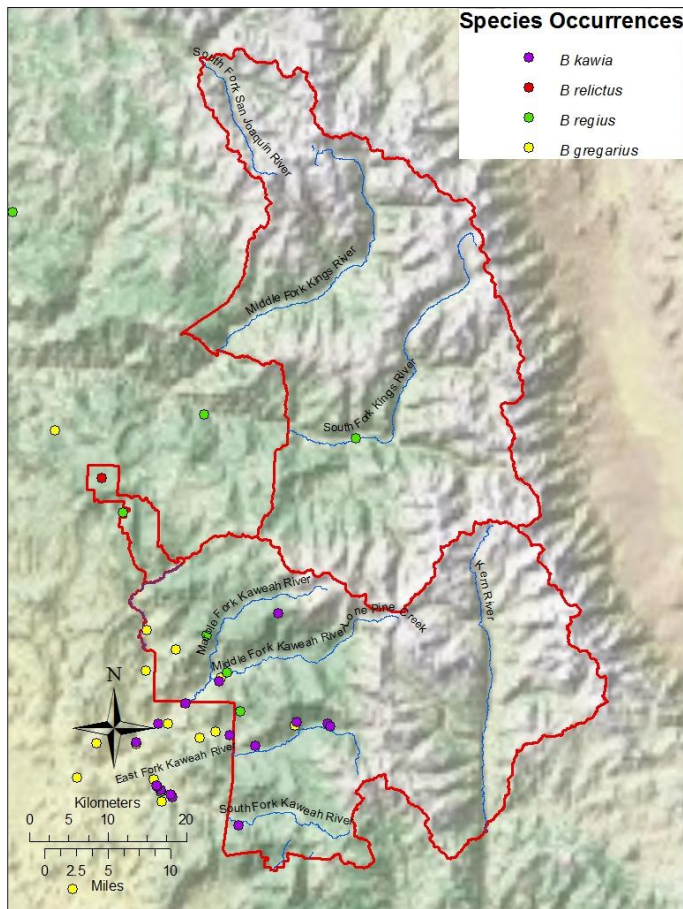
The “lungless salamanders” of the family Plethodontidae are widely distributed across eastern and western North America and Central America, with populations also in South America, southern Europe, Sardinia, and Korea. The family Plethodontidae is the largest salamander family with about 376 recognized species, about 300 of which are described to be within the subfamily Hemidactyliinae.

The slender salamanders, *Batrachoseps*, range from the Columbia River in northern Oregon (45°33') to the vicinity of El Rosario Baja California Norte (30°00') (Wake et al. 2002). The genus was once thought to contain only two species that exhibited wide intraspecific morphological variation. Currently, 20 different species are recognized within the genus *Batrachoseps*, 19 of which exist only within the state of California. However, the phylogenetics of the genus *Batrachoseps* are still a point of active research and should be treated with caution. Population fragmentation resulting from a dynamic geological landscape, divergence in allopatry, and recontact zones are responsible for the radiation within the genus and the existence of genetically distinct, range-restricted populations (Jockusch and Wake 2002). For example, the species once identified as *B. relictus* was split into four species (listed from north to south in their distribution across the Sierra Nevada): *B. diabolicus*, *B. regius*, *B. kawia*, and *B. relictus*. Sequoia and Kings Canyon National Parks houses four species of the genus *Batrachoseps*: *B. regius*, *B. gregarius*, *B. kawia*, and *B. relictus*. All species are widespread in the lower elevations (<6,000 ft) in Sequoia and Kings Canyon National Parks (David Wake, personal communication). Herein we focus on two species of interest to the National Park Service: *B. kawia* and *B. regius*.

### **Distribution**

*B. kawia* (*the Sequoia slender salamander*). The entire known distribution of *B. kawia* occurs within the boundary of Sequoia and Kings Canyon National Parks and its range is restricted almost entirely to the Kaweah River drainage at elevations of 430-2,205 m (Figure B1.1; Jockusch et al. 1998). The six confirmed sites where *B. kawia* can be found are located within the South, East, and Middle Fork Kaweah River drainages. *B. kawia* occurs in sympatry with *B. gregarius* within the Kaweah river drainage (Hansen and Wake 2005). Recent mtDNA studies indicate a sister-taxon relationship between *B. kawia* and *B. relictus* (Jockusch et al 1998). Therefore, at high elevations it is expected that *B. kawia* occurs with *B. relictus*, but no case of sympatry has been identified (Hansen and Wake 2005).

*B. regius* (*Kings River slender salamander*). *Batrachoseps regius* occupies a geographically small range and populations have been confirmed in only three areas, one of which (located along the lower Kings River) consists of a cluster of several sites (Figure B1.1). The low elevation populations within the Kings River drainage occur on the North Fork at elevations of 335-440 m (Jockusch et al. 1998). A second population, provisionally assigned to *B. regius* has been found at an elevation of 2,470 m along the South Fork Kings River drainage. More recently, salamanders discovered in the Middle Fork Kaweah River drainage at 610 m within Sequoia and Kings Canyon National Parks have been classified as *B. regius* (Jockusch and Wake 2002). Each regional group appears to be quite localized, and the degree of genetic subdivision suggests that they have been isolated from one another for a long time (Jockusch and Wake 2002). All known populations of *B. regius* occur on public lands managed by the USDA Forest Service or National Park Service.



**Figure B1.1.** Locations of *Batrachoseps* in and around the park

### **Population Biology**

The IUCN (red list) Status has listed *B. kawia* as “Data Deficient” and *B. regius* as “Vulnerable”. There is a clear lack of data regarding population trends and regional distribution for both species. Up-to date, comprehensive surveys have not been completed and historical data are lacking. Anecdotal comments suggest that additional field work will likely reveal populations of *B. kawia* in the North and Marble Forks of the Kaweah River and that more extensive surveys of the Kings River drainage will yield additional populations of *B. regius* (Hansen and Wake 2005). Two high-elevation populations of *B. kawia*, first discovered in 1982, were revisited 18 years later, but salamanders were not found (R.W. Hansen, unpublished data). The low elevation populations of *B. regius* appear to be stable as populations have been found intermittently over the past 25 years. However, these same populations of *B. regius* occupy habitat immediately adjacent to roads, and thus populations should be considered vulnerable to anthropogenically caused mortality and habitat alteration. A total of seven specimens of *B. regius* have been found at the single high elevation site on two occasions over a 45 yr period (Hansen and Wake 2005).

### **Threats**

**Disease.** Chytridiomycosis is a fungal pathogen that has caused the rapid declines of amphibian populations across the globe. While the fungus has not been identified on any individuals of *B. kawia* and *B. regius*, wild populations of *B. attenuatus* were found to be infected the disease and the fungus has been present in wild populations since 1973. Lab experiments showed that *B. attenuatus* was able to recover from infection in dry microhabitats (Weinstien 2009). While the vulnerability of a related species to chytridiomycosis should raise concerns for *B. kawia* and *B. regius* populations, the ability of the related species to recover from infection suggests that even if the species are exposed to the disease, exposure may not lead to catastrophic die-offs.

**Climate Change.** While no studies have projected the effects of climate change on either *B. kawia* or *B. regius*, it is reasonable to expect that populations of *B. kawia* and *B. regius* will not be impacted by local changes in climate. The phylogeography of *B. kawia* and *B. regius* indicates that both species have survived historic climate oscillations of the scale of current climate change projections. Both *B. kawia* and *B. regius* appear to have very restricted ranges comprised of disjunct, localized populations. The degree of genetic subdivision within populations of *B. kawia* suggests that it has limited ability for dispersal.

### **Future Action**

While information is lacking regarding the diversity, species range, and population dynamics of species within the genus *Batrachoseps*, it is reasonable to assume that populations of *B. kawia* and *B. regius* are not in immediate danger of extinction (David Wake, personal communication). The absence of information suggests that active monitoring by the NPS is needed. Such endeavors should be undertaken with caution because the cryptic nature of these species makes identification based on phenotypic traits challenging and inaccurate. As individual species ranges are of limited size, future population declines can be best averted by preserving existing habitat and preventing further alteration by the creation of roads or asphalt trails (David Wake, personal communication). Severe alteration of relatively small areas of habitat could potentially drastically reduce the species diversity of the genus *Batrachoseps*.

## B2. Sierra Nevada Chipmunks—*Tamias* spp.

### Abstract

In addition to the species identified for assessment by park resource biologists, we provide a brief assessment of the species in genus *Tamias* (chipmunks). Several species are native to the parks. These species are globally secure, but regional populations may be at risk.

- **Global status:** G5. Species is secure.
- **Park status:** Uncertain, but preliminary evidence suggests species declines
- **Overall integrity:** Moderate
- **Certainty of integrity:** Low
- **Metric to evaluate integrity:** Grinnell resurvey project
- **Key Vulnerabilities:** unknown.

### Taxonomy and General Background

Three species of chipmunk are recognized as native to Sequoia and Kings Canyon National Parks (SEKI) with an additional two species of uncertain or problematic status (Table B2.1). Of these five chipmunk species *Tamias alpinus* and *T. umbrinus* are recognized as possibly vulnerable within the state of California, while two subspecies of *T. speciosus* are considered imperiled or critically imperiled (NatureServe 2010). However, the conservation statuses of these species were assessed without the benefit of recent mammalian surveys in the Sierra Nevada and updates are likely necessary.

**Table B2.1.** Chipmunk species of Sequoia and Kings Canyon National Parks. NatureServe conservation statuses at the global (G), state (S) and/or subspecies (T) levels are provided as well as the species occurrence status within SEKI. Conservation rankings are defined as Secure (5), Apparently Secure (4), Vulnerable (3), Imperiled (2), and Critically Imperiled (1) (NPS ; NatureServe 2010).

Species		Conservation Status		
Scientific Name	Common Name	Global	State	SEKI Status*
<i>Tamias alpinus</i>	Alpine Chipmunk	G4	S3-4	Native
<i>Tamias merriami</i>	Merriam Chipmunk	G5	S5	Native
<i>Tamias speciosus</i>	Lodgepole Chipmunk	G4	S4	Native
<i>T. s. callipeplus</i>	Mt. Pinos Lodgepole Chipmunk	T1		
<i>T. s. speciosus</i>	Lodgepole Chipmunk	T2		
<i>Tamias senex</i>	Shadow chipmunk	G5	S5	Uncertain
<i>Tamias umbrinus</i>	Uinta or Inyo chipmunk	G5	S3-4	Uncertain

\* Subspecies of *T. speciosus* were not represented in the SEKI vertebrate checklist (NPS)

### The Grinnell Resurvey Project

The response of Sierra Nevada mammal species to climate change has been given a large amount of attention, in large part due to recent resurveys of the Grinnell/Storer transects near Yosemite National Park (YOSE) and elsewhere. Results from these resurveys have shown a dramatic upward shift of many species, likely linked to warming temperatures attributed to recent climate change (Patton, Wagtendonk et al. circa 2010). In fact the elevation range limits of half of the 28 mammalian species surveyed showed an average upward shift of approximately 500 meters over a period corresponding to a ~3°C increase in minimum temperatures (Moritz, Patton et al. 2008).

Moritz et al. (2008) documents the range shifts of four Sierran chipmunk species, three of which are known to occur within SEKI as well. Of these four species, *T. alpinus* and *T. senex* both showed significant range contraction, while *T. quadrimaculatus* and *T. speciosus* showed little change in their range limits (Table B2.2., Moritz, Patton et al. 2008). Modeling work building on the findings by Moritz et al. (2008) suggests that the range of *T. alpinus* is strongly limited by temperature, leading to its apparent range collapse (Rubidge, Monahan et al. 2011). The range collapse of *T. senex* can be better explained by changes in both climate and vegetation over the last century and an apparent increase in the abundance of *T. speciosus* in YOSE can be attributed to reduced congener competition as *T. alpinus* and *T. senex* populations were reduced (Rubidge, Monahan et al. 2011).

Also noteworthy is the appearance of *T. amoenus* (Yellow-pine chipmunk), which was not observed in YOSE previously. These observations indicate an approximately 500 m expansion of the species' upper elevation limit within the past three decades (Patton, Wagtendonk et al. circa 2010).

**Table B2.2.** Elevational range shifts observed along the Grinnell Yosemite resurvey transect for four chipmunk species. Given is the original elevation range during the Grinnell era and the observed change of the species lower (L) and upper (U) range limits. Table adapted from Moritz et al. 2008.

Species	Original Elevation Range (m)	Elevation Range Change (m)
<i>Tamias alpinus</i>	2307 – 3353	+629 L
<i>Tamias quadrimaculatus</i> *	1494 – 2210	+50 U
<i>Tamias senex</i>	1402 – 2743	+1007 L, –334 U
<i>Tamias speciosus</i>	1768 – 3155	+128 L, +65 U

\*It is unlikely that *T. quadrimaculatus* occurs within SEKI (NPS).

### **Sequoia and Kings Canyon National Parks**

Resurvey efforts focused in and around SEKI had not yet been published at the time of this report, but preliminary findings show an apparent loss of *T. umbrinus* from the Southern Sierra Nevada. Historic surveys conducted by Grinnell and Museum of Vertebrate Zoology (MVZ) staff produced 31 specimens of *T. umbrinus* in the Southern Sierra Nevada, between Kearsarge Pass and Cirque Peak from August 1911 to June 1912 (MVZ specimen database, <http://arctos.database.museum/>, 24 March 2011). Surveys conducted from 2007 to 2010 by Jim Patton and other MVZ Staff failed to record any evidence of this species at these sites (Patton 2011). Causes of *T. umbrinus*' disappearance are not yet clear. Likewise it remains to be seen if *T. alpinus*, *T. senex*, and *T. speciosus* exhibit similar range shifts in SEKI in response to changing environmental conditions as Moritz et al. (2008) showed in YOSE. Look for future publications on the subject from the Grinnell Resurvey Project (<http://mvz.berkeley.edu/Grinnell/index.html>).

Management efforts should undertake the task better surveying, and determining a baseline for these species in terms of distribution, diversity, population structure and habitat relations within the park. Principle threats may be vegetation change through fire and climate change.





## C. Sensitive Fishes, Native and Introduced.

### Abstract

There are three native and several non-native salmonid fishes found in the park. The three native taxa are in very poor condition. The non-native taxa have a variety of impacts on biological diversity. Management of these introduced fishes is made difficult by their widespread presence, reproductive success, and the social popularity of fish in montane lakes.

### Native taxa

- **Species global status:** *Oncorhynchus mykiss whitei*: G5T2Q. *Onchorynchus mykiss gilberti*: G5T1Q. *Oncorhynchus mykiss irideus*: G5T4
- **Park status:** Rare to non-extant.
- **Overall integrity:** Very low throughout ranges of all three taxa.
- **Certainty of integrity:** High. Significant sampling efforts find few intact populations.
- **Metric to evaluate integrity:** Field surveys.
- **Vulnerability with respect to key stressors:** Introgression with non-native fishes and habitat degradation are the primary drivers.

### Non-native taxa

- **Degree of invasion:** Extensive at high and mid-elevation lakes and low elevation streams
- **Degree of invasiveness:** High
- **Impacts in invaded habitats:** Significant
- **Control status / recommendations:** Fish removal programs are well tested in the park and are underway.

### Salmonid Taxonomy

The fish of the family Salmonidae are generally fusiform in shape, with a dorsal adipose fin and a forked tail (Moyle 2002). The juveniles often display dark vertical bands known as parr marks, though these are sometimes retained in the adult forms. The family is comprised primarily of various forms of trout and salmon. These commonly used terms have little phylogenetic meaning, since both trout and salmon native to the Atlantic share the genus *Salmo*, and the trout and salmon of the Pacific share the genus *Oncorhynchus* (Stearley and Smith 1993). In this report, we are concerned only with trout of the genera *Oncorhynchus*, *Salmo*, and *Salvelinus*, the only salmonids found within the boundaries of SEKI. *Oncorhynchus* are characterized by having only black spots on their body, while *Salmo* have black or reddish-orange spots, and *Salvelinus* generally have lighter colored spots (Behnke and Tomelleri 2002).

California is home to the widest diversity of native freshwater trout of any U.S. state, all in the genus *Oncorhynchus*, along with a wide array of introduced trout (Moyle 2002). Given our focus on SEKI, however, the categories that we will refer to as “native” and “non-native” are determined with respect to the boundary of Sequoia and Kings Canyon National Parks, not California. As such, the native trout are the Little Kern golden (*O. mykiss whitei*), endemic to the Little Kern River and its tributaries, the Kern River rainbow (*O. m. gilberti*), endemic to the upper Kern River and its tributaries, and the coastal rainbow (*O. m. irideus*; hereafter “rainbow trout”), native to lower elevations of the Kaweah and Kings Rivers and their tributaries. The non-native trout include eastern brook trout (*Salvelinus fontinalis*; hereafter “brook trout”), brown

trout (*Salmo trutta*), and California golden trout (*O. m. aguabonita*). Rainbow trout includes the common coastal form *O. m. irideus* as well as multiple hatchery-derived strains, many of which contain subspecies of unknown origin. Rainbow trout have also been stocked into many SEKI waters that are outside of their native range. Brook trout is native to eastern North America and brown trout is native to Europe, western Asia, and northern Africa. California golden trout, most closely related to the Little Kern golden and Kern River rainbow trout, are native to an area just outside the SEKI boundary, but have been stocked into many waters within SEKI. Given California golden trout's threatened conservation status, however, some of these isolated non-native populations may have conservation value, an issue which is addressed in the "Stewardship" section of this report.

### **Salmonid Life History**

The primary life history characteristics of the trout within SEKI, including habitat, feeding, reproduction and survival, and movement and interactions (all described in more detail below) are relatively similar between species and subspecies, with some notable differences. All of the fishes described above have been studied quite extensively, with the exception of the Little Kern golden trout and Kern River rainbow trout, which are less well studied but are both presumed to be similar to California golden trout (Moyle 2002). For that reason, "golden trout" in this section will refer to California golden, Little Kern golden and Kern River rainbow trout.

*Habitat.* In general, the trout in SEKI are adapted for cold, clear water, with optimal temperatures in the range of approximately 14-20 °C. With proper acclimation, trout can survive in water up to about 26 °C, but growth is limited in such situations (Moyle 2002). Brown trout are the most heat tolerant, able to survive at 29 °C for short periods. Brook trout are likely the most cold tolerant, and will feed in water as low as 1 °C. All of these trout can live in streams or lakes, though in streams they do best when there is in-stream cover available, and a variety of runs, riffles and pools. Suitable gravel for breeding can be limiting, particularly in smaller streams, and therefore becomes an important habitat component for fish at higher elevations near headwaters (Behnke 1992; Knapp et al. 1998). Brook trout will readily breed in either lakes or streams, while rainbow and brown trout usually breed in streams, even if they later reside in lakes. Golden trout seem to require access to stream habitat for breeding (Moyle 2002), and breeding success is generally limited when stocked into lakes without such access. In addition to breeding habitat, trout also require rearing habitat, foraging habitat, and over-wintering habitat. These are usually deeper water habitats with protective cover, and any of these habitats can limit populations if they are in short supply (Behnke 1992).

*Feeding.* SEKI's trout are generalists that feed primarily on invertebrates, which can include benthic and pelagic aquatic organisms, or terrestrial organisms (Moyle 2002). Brown and brook trout will also eat fish when they attain larger sizes, and can be important predators in freshwater systems. All of these trout will aggressively defend feeding territories in lotic systems, especially in faster running water, which can lead to competition where good feeding habitat is limited (Behnke 1992).

*Reproduction and survival.* The timing of reproductive events is the most notable difference between these trout species. Brook, brown, and rainbow trout all mature in their second or third year, while golden trout are slower to mature, usually in their third or fourth year (Moyle 2002).

The breeding season is also variable, with brook and brown trout breeding in fall or winter, and rainbow and golden trout breeding in spring or summer (Behnke 1992). Golden and rainbow trout's shared breeding season and genetic similarity leads to hybridization and introgression, which is not a concern with brook or brown trout. The earlier spawning of brook and brown trout relative to golden, however, can lead to significant size difference between the juveniles of the different species, and result in competitive advantages or predation.

The trout of SEKI tend to live approximately 6–7 years (Behnke, 2002; Moyle 2002), although there are cases of trout living to 20 years or longer in the Sierra Nevada (D. Boiano, pers. comm.). Golden trout, in particular, has been shown to have a relatively slow growth rate, which tends to be negatively density dependent (Knapp 1996).

*Salmonid interactions.* In many aquatic systems, particularly larger rivers, multiple salmonids coexist successfully. Coexistence often involves greater niche specialization and differentiation by the two species, thereby minimizing direct competition (Behnke 1992). In the small lakes and streams of SEKI's higher elevations, however, it is rare to find two sympatric forms, often to the detriment of the native fishes. Brook and brown trout are usually stronger competitors than golden trout, and will prevent golden trout's establishment in areas where either brook or brown trout is present. Rainbow trout and golden trout do not appear to compete with each other in this way, but will readily interbreed and create viable hybrid offspring, which also results in the loss of the native forms (Behnke 1992).

## **C1. Native Trout**

### ***The Golden Trout Complex, *Oncorhynchus mykiss*.***

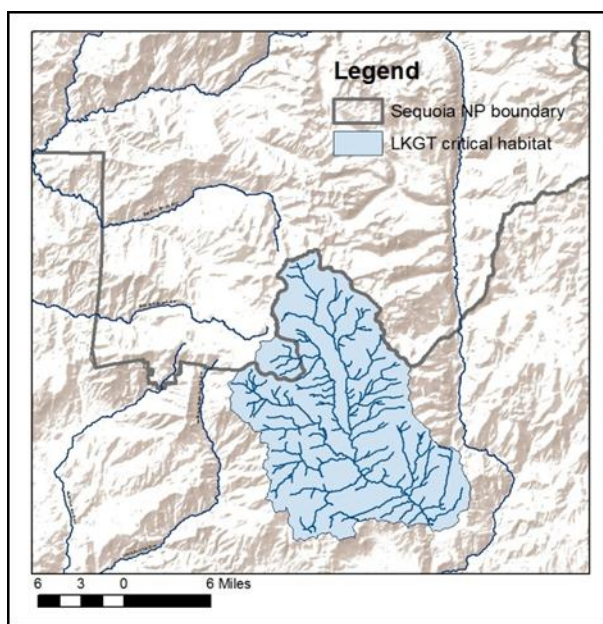
*General Distribution.* *O. mykiss* has diversified into numerous subspecies in the many isolated watersheds found throughout the state. Two of these *O. mykiss* subspecies are native to selected mid-elevation areas within SEKI, including the Little Kern golden and Kern River rainbow, native to the Little Kern River drainage and Kern River drainage, respectively. Along with California golden trout, native to Golden Trout Creek and the South Fork Kern River outside SEKI, these three subspecies are known as the "golden trout complex." Although there has been extensive debate about their correct classification, they are now well established to be distinct subspecies of *O. mykiss*. Their exact phylogenetic relationships, however, including their relationship to other *O. mykiss* forms, have not been firmly established. A third *O. mykiss* subspecies, the coastal rainbow, is also native within SEKI and is found in the lower elevation stream reaches of the Kings and Kaweah Rivers. However, this latter subspecies is not included as a native species in this report due to its ubiquitous distribution throughout the state, and is addressed as a non-native trout in the non-native trout section of this report due to its history of being stocked in non-native waters within SEKI.

### ***Little Kern Golden Trout. (*O. mykiss whitei*)***

*Distribution within SEKI.* Little Kern golden trout are only found within SEKI in the upper tributaries of the Little Kern River, notably Upper Soda Spring Creek (Figure C1.1). This habitat is in good condition for Little Kern golden trout (Trout Unlimited 2009). Accurate estimates of population connectivity between the SEKI habitat and downstream habitats outside of the park are not available. A fish barrier was constructed in 1971 to prevent upstream migration of non-

native fish into Upper Soda Spring Creek, so it is likely that downstream migration is far more successful than upstream, and upstream migration is likely not possible. Given the tendency of golden trout to move very little within a stream (Matthews 1996), it is possible that there was limited connectivity between the SEKI tributaries and other parts of the range, even before the placement of the fish barrier.

The headwaters of the Little Kern, often above natural barriers, were likely fishless historically, but human introductions of fish above these barriers has made it difficult to determine the precise historic range. Little Kern golden trout were also transplanted from the Little Kern basin into several nearby lakes and streams, including Coyote Creek, a tributary of the Kern River to the east (Molly Stephens, pers comm.), though these populations have hybridized with rainbow trout and thus no longer represent pure examples of the subspecies.



**Figure C1.1.** The Little Kern River and its tributaries, with Little Kern golden trout critical habitat shaded in blue. SEKI lies north of the gray border, indicating the small amount of Little Kern golden trout habitat within the park boundary.

### Threats

The primary threat to Little Kern golden trout is hybridization with introduced rainbow and California golden trout, present due to fish stocking in the region (Christensen 1984). Habitat degradation, primarily caused by cattle grazing, has also been a factor in some areas of the Little Kern golden trout's range, but not within SEKI. The negative effects of grazing include increased erosion and the resulting loss of suitable spawning substrate, as well as reduction of desirable aquatic vegetation which the trout use for cover (Knapp et al. 1998). Both of these threats have been largely abated due to management actions over the past 30 years, but hybridization, in particular, remains a concern. The continued presence of non-native and introgressed individuals in parts of the range, along with the possibility of unauthorized fish introductions, represents an ongoing threat to the integrity of Little Kern golden trout populations (USFWS 2001). Better understanding of natural fish barriers, which can prevent the spread of

non-native genotypes, is crucial for accurately assessing this threat, although current knowledge indicates that such barriers exist for all but the lower portions of the Little Kern river (R. Knapp, pers. comm.).

Looking forward, the effects of climate change may also pose threats to Little Kern golden trout habitat. Specifically, increased risk of fire, flood, and drought could put affected populations at risk (Trout Unlimited 2009). Habitat connectivity within the subspecies' range will be important for the persistence of the fish if the predicted climate-related changes occur.

#### Populations Trends and Current Status

Little Kern golden trout abundances likely began declining in the late 1800s, when non-native fish stocking and angling started to become more common in the region. By 1971, it was estimated that pure Little Kern golden trout only existed in 5 isolated populations in less than 10 miles of stream (Christenson 1984). Accordingly, Little Kern golden trout was listed as a threatened species under the federal Endangered Species Act (ESA) in 1978 (43 FR 15427 15429). Critical habitat was designated concurrently with its listing, and includes the entire Little Kern River drainage above a natural fish barrier that lies below the southernmost tributary. This area includes the entire Little Kern golden trout habitat within SEKI.

Active management for Little Kern golden trout, beginning in 1975, has restored Little Kern golden trout to nearly 100 miles of streams within its historic range. Recent genetic evidence indicates that much of this habitat now contains pure Little Kern golden trout (Stephens 2007), including all populations located within the SEKI park boundary, primarily in Upper Soda Spring Creek. Introgressed individuals remain in several populations within the Little Kern drainage, however, particularly in the middle and lower sections of the mainstem Little Kern River. This continued threat of hybridization is likely the reason NatureServe still lists Little Kern golden trout as "declining." The USFWS and CDFG, however, both estimate that the overall health of the subspecies is greatly improved since its listing in 1978, and are now considering it for delisting. Likewise, Trout Unlimited's Conservation Success Index (2009), a composite measure of subspecies condition based on several data sources, gives Little Kern golden trout high scores throughout most of its range.

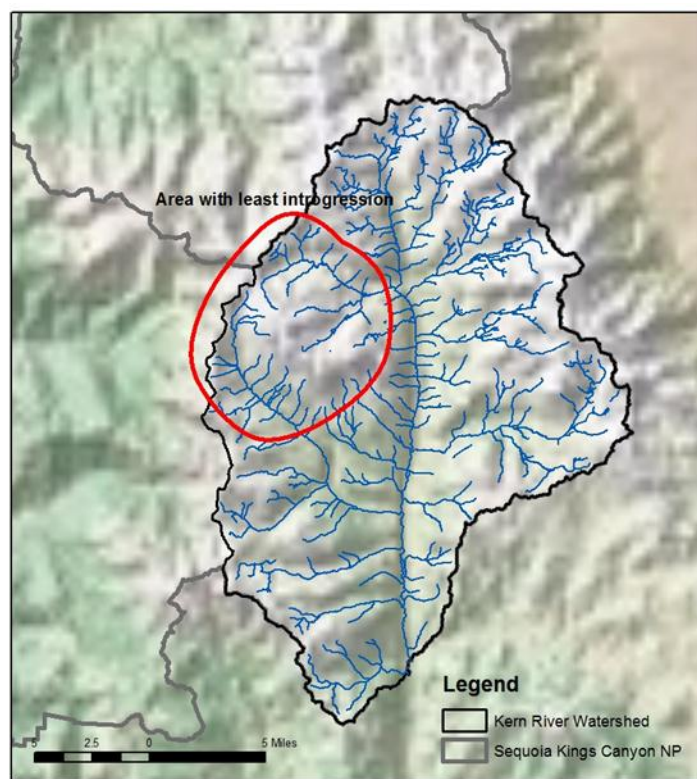
The Little Kern golden trout within SEKI are reported to show no introgression, and therefore represent the desired outcome of management efforts. Continued protection and monitoring within SEKI are important to ensure this section of the range and the integrity of the fish remains intact. SEKI encompasses only a small section of the Little Kern golden trout range, so most management will take place outside of the park boundary. Nonetheless, the SEKI populations are an important part of the re-establishment of the subspecies, and are often used as the genetic standard to which putative Little Kern golden trout are compared in other sections of the range (Molly Stephens, pers. comm.).

#### ***Kern River rainbow trout (O. mykiss gilberti)***

*Distribution within SEKI.* Kern River rainbow trout are endemic to the Kern River basin, and occupy the mainstem Kern River and its tributaries (Figure C1.2). Although the precise historic range remains unknown, several sources cite Junction Meadow as the northernmost reach of the Kern River rainbow's range, since areas above that were likely glaciated and therefore fishless (Boiano et al. 2005). Sumner (1936) noted that the Kern River above Junction Meadow, and

Tyndall Creek (near the Kern River headwaters), were both “plentifully supplied with the Kern Rainbow.” This statement, however, likely reflects misidentification of stocked fish, since these areas were both commonly stocked with non-native trout beginning in the late 1800s.

The extensive stocking of the region, combined with the propensity of Kern River rainbows to hybridize with stocked non-native rainbow and California golden trout, led to skepticism about the continued existence of the fish (Schreck 1969, Moyle 2002). Genetic studies beginning in the 1970s have attempted to assess the extent to which pure Kern River rainbow trout might exist, but this process is made difficult by the lack of a known type specimen of Kern River rainbow trout for comparison. It has, nonetheless, been determined that fish below Durwood Creek (just south of the SEKI boundary) are almost entirely non-native rainbow trout. Similarly, the headwater tributaries of the Kern River are populated with introduced California golden trout, or hybrids between them, rainbow trout and Kern River rainbows. In between these two sections, however, there are isolated areas where Kern River rainbow were stocked outside of their native range, in which fish with native genotypes appear to exist. Bagley (1997) found genetic evidence that Kern River rainbow might be best represented by fish in tributaries between the headwaters and the entrance of Golden Trout Creek at SEKI’s southern border. More recent evidence confirms this result, showing populations with little introgression in Picket Creek, Kern-Kaweah River, Nine Lakes North, and Rattlesnake Creek (Erickson et al. 2009). Further genetic studies are needed to fully quantify levels of introgression and the possible extent of non-introgressed populations.



**Figure C1.2.** The Kern River basin within the SEKI boundary, with major tributaries shown. Kern River rainbow trout populations showing the least introgression exist within the circled area, with much of the rest of the watershed occupied by trout introgressed with non-native rainbow trout or golden trout.

Much of the habitat believed to contain relatively-pure Kern River rainbow trout is found within SEKI. Levels of connectivity between the various tributaries in this section have not been well established. The mainstem Kern River, however, allows for high connectivity between fish within SEKI and those further south in Sequoia National Forest. Although there are fish within the first 5 miles south of the park border which appear to have low to moderate levels of introgression, levels of introgression appear to rise towards Durwood Creek, below which there are likely no native fish extant. Perhaps of more concern is the possibility of the downstream migration of hybridized fish from the Kern River headwaters. The upper mainstem Kern River is likely the most vulnerable, since any downstream migration would reach this area first. Tributaries that enter below this upper section may be partially or entirely protected from migration into them by natural fish barriers, although the efficacy of such barriers has not been analyzed. Data are lacking to make accurate estimates of population connectivity.

### Threats

The main threats facing Kern River rainbow trout are similar to those for Little Kern golden trout; specifically, hybridization with introduced rainbow and California golden trout (Bagley 1997). The presence of these non-native trout in both the lower and the upper sections of the Kern River, as well as in many of its tributaries, makes this threat ubiquitous and difficult to address. The possibility of the downstream spread of introgression is of particular concern, since upstream migration is more likely to be impeded by natural barriers.

Habitat degradation is also a concern for Kern River rainbow trout. Beaver (*Castor canadensis*) are thought to occur within SEKI in the lower mainstem Kern River and can significantly alter trout habitat (Danny Boiano, pers. comm.), although any Kern River rainbow within this section are likely introgressed. The most likely impact of these alterations would be the decrease in available spawning grounds through the disruption of natural flow regimes. The specific effects of beavers on trout populations, however, have not been fully determined.

Kern River rainbow trout is likely to experience the same effects of climate change as Little Kern golden trout, namely increased likelihood of flood, fire, and drought. With a larger potential range than Little Kern golden trout, Kern River rainbow may have a higher capacity to persist in the face of climate change, especially if the previously listed threats can be minimized.

### Population Trends and Current Status

Historic population trends are difficult to establish for Kern River rainbow trout due to hybridization, which makes accurate identification of pure Kern River rainbow trout very difficult. This extensive hybridization was compounded by habitat degradation outside of SEKI, including the Flat Fire in 1976 and resulting landslides, which may have negatively impacted breeding habitat. A 1992 survey by the CDFG found approximately 1000 trout per mile in a survey of the Kern River within SEKI, but was unable to determine with any certainty the introgression status of these fish. Due to concerns about the Kern River rainbow's continued existence, it was listed as a Species of Special Concern in California, and as a Category 2 taxa under the federal ESA, indicating that listing may be appropriate but that sufficient data are lacking (1994: 59 FR 58982 59028). NatureServe lists the Kern River rainbow as "critically imperiled," acknowledging that no accurate population estimates exist for pure Kern River rainbow.

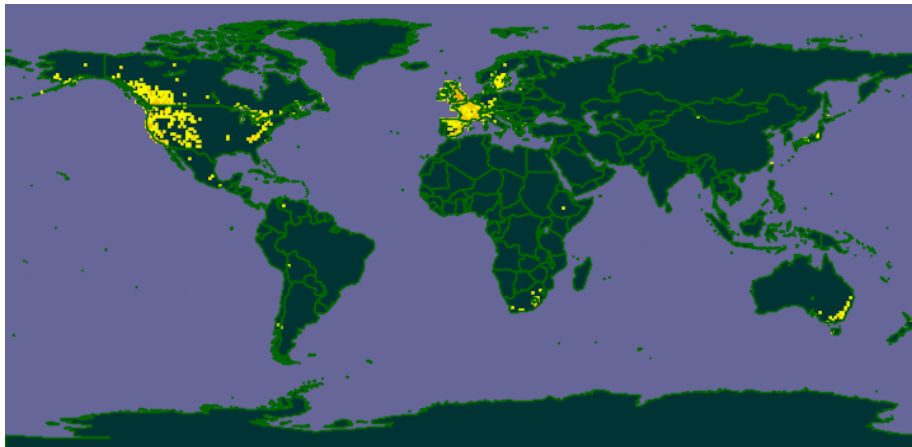


SEKI encompasses the majority of the habitat thought to still contain Kern River rainbow trout. As such, SEKI has been an active participant along with CDFG and Sequoia National Forest in the Upper Kern Basin Fishery Management Plan (1995, currently under revision).

## C2. Trout Introduced to Sequoia and Kings Canyon National Parks

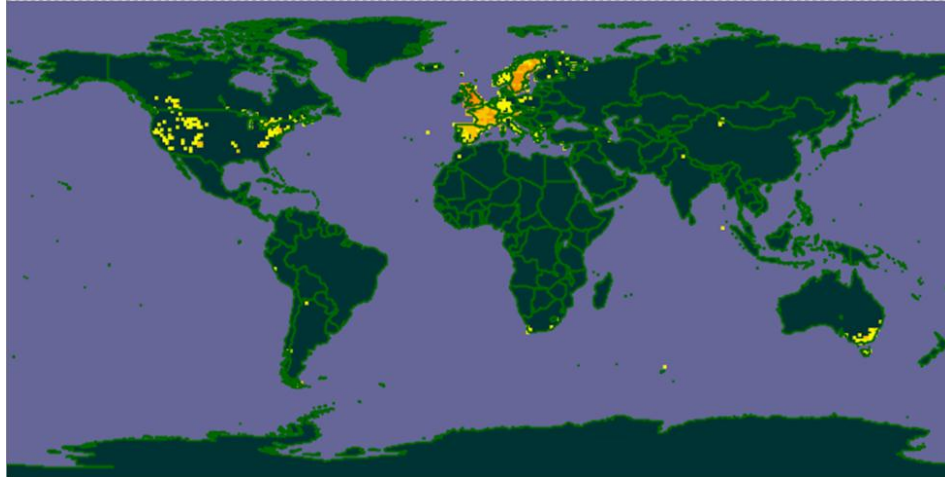
### **Global Distribution**

The genera *Oncorhynchus*, *Salmo*, and *Salvelinus* form a diverse group of fish that have been successful over an enormous geographic area spanning every continent except Antarctica. This global range is largely due to their longstanding popularity for both sport and commercial fishing, and the resulting introduction into many areas outside their native ranges (Pister 2001). *O. mykiss* has been introduced into 38 countries (Figure C1.3), *S. trutta* has been introduced into 28 countries (Figure C1.4), and *S. fontinalis* has been introduced into 18 countries (distribution data from Global Biodiversity Information Facility-GBIF). No map is available for the worldwide distribution of *S. fontinalis*. Non-native trout have impacted low-elevation stream systems by causing hybridization and competition with native fishes. When introduced into historically fishless water bodies at high elevations, the completely novel predation pressures have altered entire food webs (Knapp et al 2001). Thus, the result of worldwide stocking of trout has led to dramatic impacts to ecosystems around the world (Gozlan et al 2010).



**Figure C1.3.** Worldwide distribution of *Oncorhynchus mykiss*. This species is native to the Pacific basin, from Japan across the north rim and south to Mexico. Of the current distribution in 44 countries, its native range extends into only 6. The remaining populations are the result of human introductions (map from GBIF).





**Figure C1.4.** Worldwide distribution of *Salmo trutta*. This species is only native to the northeastern Atlantic. Of the current distribution in 57 countries, its native range extends into only 29. The remaining populations are the result of human introductions (map from GBIF).

#### Regional Distribution

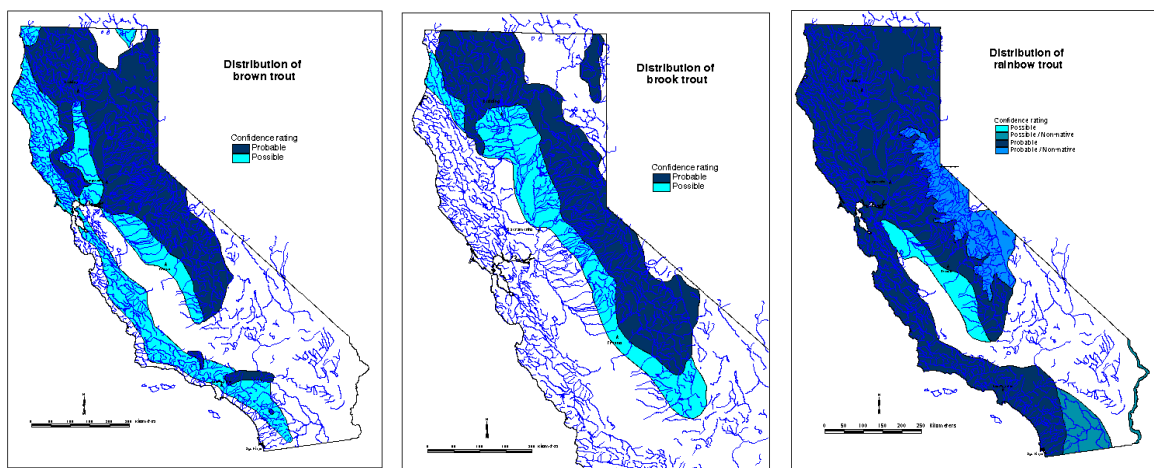
In California, extensive introductions of non-indigenous fish species have occurred since the later part of the 19<sup>th</sup> century (Pister 2001, Moyle 2002). This includes species not native to California, such as the brook and brown trout, as well as California natives that have been moved to water bodies outside their historic range (e.g., rainbow trout subspecies of mixed and unknown origin). Early stocking was mostly conducted by private citizens and groups, but by the 1920s the California Department of Fish and Game was planting fish on a regular basis (Pister 2001). By the 1960s, with the advent of the environmental movement and increased awareness of the impact of non-native fish on ecosystems, official stocking of new water bodies began to decrease.

The National Park Service set a policy against stocking in 1972 (NPS 1975), with limited plantings continuing in Sequoia, Kings Canyon, and Yosemite National Parks until 1991 (Moyle et al 1996). Today, the CDFG continues to stock trout on national forest lands in the Sierra Nevada to maintain fish populations for sport fishing, under the assumption that the current populations are not capable of sustaining themselves. Due to the large number of water-bodies in the Sierra Nevada, surveys of biodiversity and records of trout populations are incomplete. We do know, however, that all major watersheds of the mountain range contain introduced trout populations (Figure C1.5, Moyle et al 1996).



**Figure C1.5.** Adapted from Moyle et al (1996). Shaded area is that believed to be largely without fish before human introductions.

These non-native trout can also be found in many of California's low to middle elevation reservoirs filled by Sierra Nevada and Coast Range streams and rivers (Figure C1.6). Some of these streams and rivers have been managed specifically as brown trout streams as part of the California Wild Trout Program. Below we describe current distribution and populations trends separated into two categories (1) introduced trout in high elevation (>2500m) water bodies and (2) introduced trout in low (<1000m) elevation and mid (1500m – 2500m) elevation water bodies.



**Figure C1.6.** Distribution of rainbow, brown and brook trout in California. Maps generated for these three species were created for The Nature Conservancy Hexagon Project, a study of California freshwater fish distributions and can be found at the UC Davis Information Center of the Environment (ICE).

Trout introduced into high elevation water bodies (>2500m)  
(Rainbow trout, California golden trout, brook trout, and brown trout)

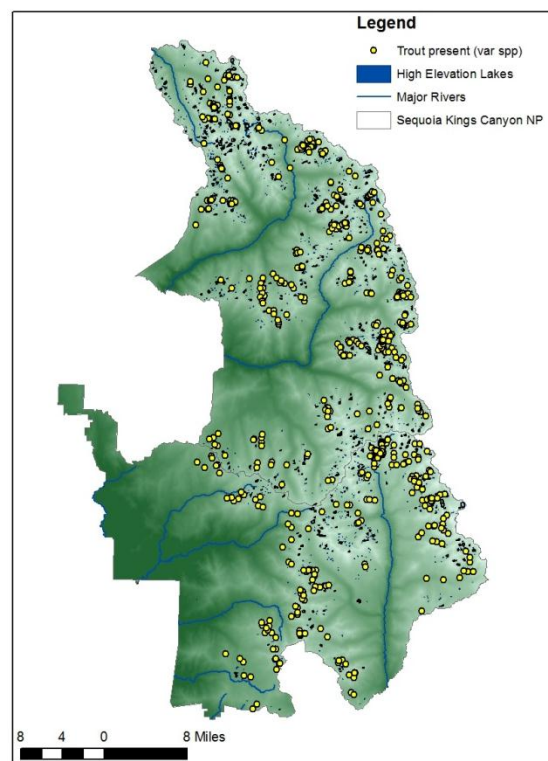
*Habitat Associations.* The high elevation lakes of the Sierra Nevada that now contain self-sustaining trout populations were carved during past glaciation events. Most are small or moderate in size (<10 ha), and widespread granitic geology contributes to oligotrophic (low productivity) conditions (Boiano et al 2005, Moyle et al 1996). They are covered with ice for up to eight months of the year, and rarely reach maximum temperatures of more than 17 °C (Bradford 1983, Bradford et al 1998). Armstrong and Knapp (2004) quantified habitat characteristics that were highly correlated with reproductive trout populations in the southern Sierra Nevada. This study concluded that trout are likely to persist in alpine lakes that contain more than 2.1 m<sup>2</sup> of spawning habitat and are lower than 3520 m in elevation. (Spawning habitat was defined as uncemented gravels between 0.5-4 cm in size, in water between 10-50 cm deep, with velocities between 20-60 cm/s.) The amount of habitat is positively correlated with the size of the outlet streams, and can vary from year to year depending upon winter precipitation (Armstrong and Knapp 2004). Stream outlets increase spawning habitat for trout, and also increase the connectivity of these high elevation systems. This connectivity in turn assists introduced trout populations in persisting despite natural stressors and intentional eradication projects.

*Population Trends and Current Status.* Trout are well adapted for the cold waters of alpine lake habitats, but they are often naturally excluded from these isolated and rugged areas due to impassible fish barriers (Moyle et al 1996). Historically, 95% of mountain lakes in North America did not contain populations of fishes (Bahls 1992). Estimates for the Sierra Nevada are similar, where it is believed that most of the lakes and streams above 1800 m were historically fishless (Knapp 1996), although trout may have naturally reached elevations up to 2200 m in the Middle Fork Kings River, 2400 m in the Kern River within SEKI, and 3000 m in the Kern River outside of SEKI (Moyle et al 1996, Moyle 2002). Historically fishless water bodies provided important habitats for native fauna. In the interest of creating recreational fisheries, however, planting began in these lakes in the mid- to late-1800s (Bahls 1992, Pister 2001). By the late 1900s, between 80-95% of fishless lakes in the Sierra Nevada that were large enough to support fish contained at least one species of introduced trout (Moyle et al 1996).

*Drivers of temporal variation (climatic, anthropogenic, biological).* The primary driver of temporal variation in high elevation trout populations is anthropogenic, including previous stocking, current eradications, and recreational angling. By the mid-1970s, most large water bodies in SEKI had been stocked at least once with non-native fish, and many had been stocked repeatedly. Trout stocking was phased out in Sierran national parks beginning in 1975 (NPS 1975, unpublished fish-stocking records from Knapp and Matthews 2000) and SEKI's last reported stocking was in 1988 (NPS unpublished records). Although stocking no longer occurs in SEKI, nonnative fish had established self-sustaining populations in approximately 573 water bodies (NPS unpublished records) and in hundreds of miles of stream. From 1997 to 2010, fish eradication projects conducted by researchers (Vredenburg 2004) and SEKI (NPS 2010) reduced the number of non-native fish populations to approximately 561 water bodies. The remaining approved restoration work will likely reduce the number of non-native fish populations to

approximately 547 water bodies. Proposals for future restoration have the potential to eventually eradicate non-native fish from up to an additional 15% of SEKI's fish-containing water bodies.

During the period of 1960-1977, 84 lakes were formally stocked. Of these, 68% still contain self-sustaining populations of trout after more than twenty years without supplemental plantings (Figure C1.7; Armstrong and Knapp 2004). Due to the decades that have passed since stocking ceased in the Sierran national parks, most remaining populations in SEKI are likely to be self-sustaining. Thus we can assume that the populations of introduced trout in these high elevations lakes and streams are stable in the absence of human intervention or shifting habitat availability due to climate change (Danny Boiano, pers comm). However, the presence of non-native trout and limited enforcement might also allow for illegal stocking of lakes within SEKI. On public lands adjacent to SEKI, levels of recreational angling demand will inform the level of formal stocking necessary to sustain trout populations.



**Figure C1.7.** Distribution of known non-native trout presence in SEKI at high elevations (unpublished data, Knapp, RA.). Yellow dots indicate trout presence and may include one or more introduced trout species.

*High Elevation (>2500m) Threats.* Non-native fishes have dramatically altered aquatic ecosystem structure (e.g., food web dynamics) and function (e.g., nutrient cycling; Ogutu-Ohwayo 1990, Finlay and Vredenburg 2007, Schindler et al. 2001). These impacts to aquatic ecosystems can also extend into adjacent terrestrial ecosystems as a consequence of changes in resource subsidies (Epanchin et al 2010, Knapp 2005, Baxter et al. 2004, Nakano and Murakami 2001).

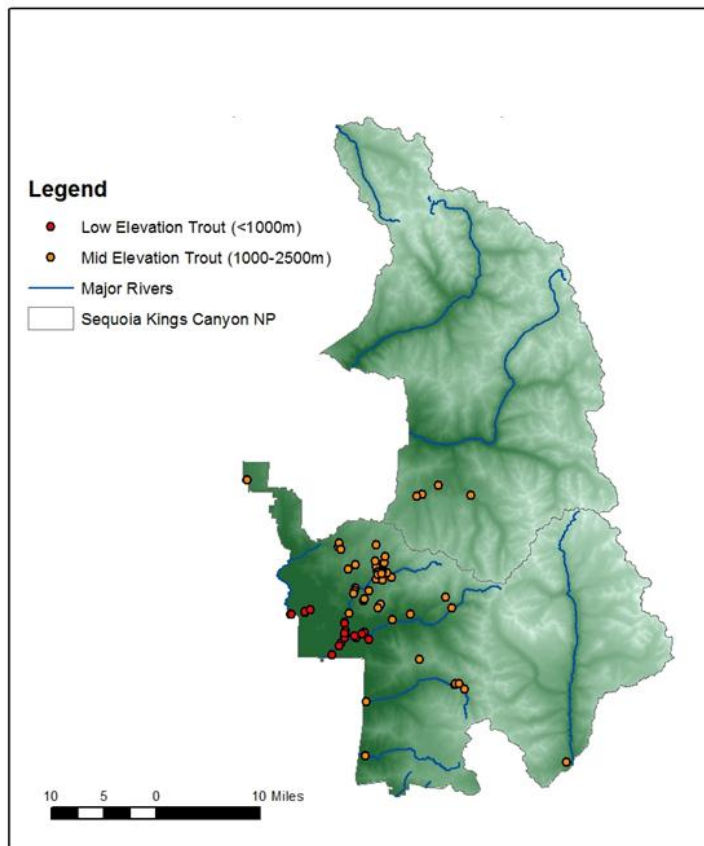
Due in part to these aquatic impacts, the Sierra Nevada Ecosystem Project (SNEP) identified aquatic habitats as one of the most altered and impaired environments in the range (SNEP 1996). Non-native trout affect a multitude of native and endemic fauna. Mountain yellow-legged frogs (*Rana muscosa*, *R. sierrae*) are particularly hard-hit species that have disappeared from more than 90% of sites in their historic range within the Sierra Nevada (Vredenburg et al. 2007). This decline appears to be due in large part to the introduction of non-native trout (Vredenburg et al. 2007). (See the mountain yellow-legged frog section of this document for further information.) Many other members of the native faunal assemblage show similar local extirpation when fish are introduced into a system, such as benthic macroinvertebrates (Knapp et al. 2001, Nystrom et al. 2001) and zooplankton (Stoddard 1987, Bradford et al. 1998, Knapp et al. 2001, Parker et al., 2001). For example, Nystrom et al. (2001) found that trout introductions into a native pond community reduced predatory macroinvertebrates by nearly 100%. Knapp et al. (2001, 2005) have also shown that large macroinvertebrates, such as mayflies (Ephemeroptera) and caddisflies (Trichoptera), as well as zooplankton, such as *Daphnia* and copepods (*Hesperodiaptomus*), are either extirpated or greatly reduced in abundance by non-native trout predation in Sierran lakes.

*Trout introduced into low (<1500m) to mid (1500m - 2500m) elevation water bodies*  
(Rainbow trout, brook trout and brown trout)

*Distribution within SEKI.* Within and adjacent to SEKI, native coastal rainbow trout are generally restricted to low elevation (< 1000 m) sites in the Kaweah River and up to mid elevations (2200 – 2400 m) in the Kings River. Native rainbow trout were restricted from reaching further upstream reaches by steep cascades. Non-native trout have been detected in multiple locations within the low (< 1000 m) and mid (> 1500 m) elevation areas in and around SEKI. Rainbow trout are currently stocked in all reservoirs downstream from SEKI drainages (CDFG Fish Planting Data).

Periodic surveys of twelve transects in the Kaweah drainage were conducted by SEKI staff from 1980 to 2007. Survey sites were broken into two elevation ranges, low (< 1000 m) and mid (> 1500 m), with six transects per elevation range. Brook trout were documented at all six mid elevation sites. Rainbow trout and rainbow-golden hybrids are the most dominant species found at each mid elevation site. Brown trout were only documented at two of the mid elevation sites, both on the Marble Fork Kaweah River. It is important to reiterate that none of the above trout including the rainbow trout are native to this mid elevation range in the Kaweah River. Brook trout were not detected at any of the low elevation sites. Brown trout and rainbow trout/hybrids were detected at all six of the low elevation sites (NPS 2011).

Formal surveys are scarce for the other low and mid elevation watersheds in SEKI resulting in relatively little data available for non-native trout. Online fishing reports indicate that non-native trout are found in many other low to mid elevation rivers. In the South Fork Kings River, brown trout have been found at Cedar Grove, Lewis Creek (which also has brook trout), Woods Creek and Bubbs Creek. Brook trout have been reported in Rattlesnake Creek and Big Arroyo Creek on the Kern River. Further reports show that both brook and brown trout are found in Kern Lake and Little Kern Lake in the lower sections of the Kern River, while brown trout have been found near Monache Meadows in the South Fork Kern River, all within the Golden Trout Wilderness adjacent to SEKI. Also adjacent to SEKI in the John Muir Wilderness, brown trout have been reported in Crown Creek and Scepter Creek ([stevenojai.tripod.com](http://stevenojai.tripod.com)) (Figure C1.8).



**Figure C1.8.** Distribution of known trout presence in SEKI at low and mid elevations. This map only contains data points with GPS coordinates. See SEKI distribution section for streams without GPS coordinates. Yellow dots indicate non-native trout presence in mid elevations and red dots indicate non-native trout species in low elevations. Each dot may include one or more introduced trout species.

*Habitat Associations.* In streams, brown trout feed in deeper pool habitat, whereas brook and rainbow trout are primarily surface feeders. Brook trout can tolerate water as cold as 1 °C, but prefer water in the range of 14-19 °C. Brown trout prefer temperatures of 12-20 °C, but tend to avoid streams that stay below 13 °C for extended periods of time. Typically brown trout have a greater tolerance for higher maximum daily stream temperatures than brook trout. Rainbow trout can tolerate intermediate ranges of temperature (Moyle 2002).

Based on data collected during the Kaweah surveys, mid elevation (> 1500 m) stream segments consist primarily of coniferous forests (*Pinus*, *Abies*) with occasional hardwoods (*Alnus*, *Populus*, *Salix*). Low elevation (< 1000 m) stream segments consist of riparian woodland vegetation (*Alnus*, *Populus*, *Salix*, and *Quercus*). Reported stream substrate data did not differentiate between elevation bands and consisted primarily of boulders, cobbles, and bedrock. Mid elevation stream composition in the Kaweah drainage was 15-57% pool (mean=38%) and 13-70% riffles (mean=40%), with the remainder being rapids. Low elevation stream composition is 27-38% pools (mean=32%) and 45-63% riffles (mean=55%), with the remainder being rapids (Werner 1981). Stream gradients ranged from 4-12 degrees in low elevation sites and from 5-25 degrees for mid elevation sites. Sinuosity ranged from 1.02-1.21 and stream development ranged

from 1.00-1.71, both using reported data that did not differentiate between low and mid elevation sites. Stream temperatures during the surveys ranged from 0.2 °C to 20.1 °C. (Werner 1986, 1991). Another study looking at both high and low elevation water bodies throughout Yosemite and SEKI found stream and lake temperature ranges of 4 °C to 17 °C, measured from May to August for elevations of 1200 m to 3200 m (Derlet and Carlson 2004). Some of the water bodies listed in the Derlet and Carlson paper overlap with the water bodies known to have nonnative trout listed above.

*Population Trends and Current Status.* While sightings of non-native trout species are found throughout SEKI, the Sierra Nevada, and the state of California (CNDDDB, SEKI published and unpublished data, online fishing reports), formal monitoring for low elevation trout species within SEKI has been limited to periodic surveys by the CDFG on the South Fork Kings River (Boiano et al. 2005) and 12 sites within the Kaweah River drainage, sampled four times since 1980 (NPS 2011, Werner 1981, 1986, 1991). As such, the Kaweah River surveys contain the richest dataset with respect to population trends in SEKI for non-native trout.

From 1980 to 2007, species composition at four of the six mid elevation sites changed very little. In 2007, rainbow trout consisted of  $\geq 94\%$  of the sample at Silliman High Remote (SHR), East Fork Mineral King (EFMK), Silliman High Near (SHN) and Marble Fork High Remote (MFHR). Brook trout made up the remainder of the sample composition in SHR, EFMK, and SHN. Brown trout and brook trout made up the remainder of the sample composition in MFHR. These results generally reflect the original surveys. Fish species composition changed much greater between sampling periods at the remaining two mid elevation sites. In Clover Creek rainbow trout decreased from 75% to 53% of the sample and brook trout increased from 25% to 47%. At Marble Fork High Near, rainbow trout decreased from 79% to 60% of the sample, while brown trout increased from 21% to 30% and brook trout increased from 0% to 10%. Population estimates ( $>110$  mm fork length) at all sites indicate that rainbow trout and brown trout are found at higher frequencies than brook trout. Mean fork length of all captures show that rainbow trout and brown trout size decreased while brook trout remained the same. In general, long-term monitoring data reveal low to moderate decreases in rainbow trout composition and increases in brown trout and brook trout composition. Mean number of captures of each species at mid elevation sites increased from 1980 to 2007. Total fish captures of all species at mid elevation sites increased from 1,019 to 3,123 fish. (NPS 2011).

From 1980 to 2007, species composition changed greatly in five of six low elevation sites. In Yucca Creek Remote, rainbow trout increased to 100% of the sample, displacing the small population of brown trout detected in prior surveys. Rainbow trout at the Middle Fork site increased, while brown trout have decreased. In North Fork Low Near (NFLN), rainbow trout have decreased in species composition. In Marble Fork Low Remote, rainbow trout decreased dramatically while brown trout increased to over half of the sample. In Yucca Creek Near (YCN), rainbow trout have exhibited consistent, robust populations. Until 2007, populations of rainbow trout at North Fork Low Remote mirrored that of YCN. However, the latest survey at this site revealed decreases in rainbow trout coinciding with a large increase in brown trout. Population estimates ( $>110$  mm fork length) reflect that brown trout are found at higher frequencies in large size classes than rainbow trout. In general, long-term monitoring data reveal low to large decreases in rainbow trout where brown trout are present. Mean number of captures



of each species in low elevation sites increased from 1980 to 2007. Total fish captures of all species at low elevation sites increased from 581 to 1,199 fish. (NPS 2011).

*Drivers of temporal variation (climatic, anthropogenic, biological).* Observations from the Kaweah surveys indicate that brook trout exist in disconnected, drying stream segments absent of rainbow trout. However, rainbow trout were found in these disconnected stream segments in subsequent surveys, suggesting that though brook trout exhibit a greater tolerance for these conditions, rainbow trout are able to repopulate the stream when conditions again become favorable (Werner 1991). It was also noted that in the Kaweah sites with high fishing catch limits, there was a high proportion of brown trout compared to rainbow trout, suspected to be due to the relative ease of angling for rainbow trout versus brown trout (Moyle 2002 p. 296, Werner 1981). This trend was not seen in low catch limit sites (Werner 1991, SEKI unpublished data). Continued recruitment of fish from high elevation source streams and lakes also supplement these low elevation streams (Moyle 2002 p. 303).

*Threats at low elevation.* A large threat from non-native salmonids in low elevation streams is the competition with, and predation of, native salmonids. Non-native salmonids may also negatively impact native herpetofauna, such as the western pond turtle and foothill yellow-legged frog (NPS 1999). Brown trout are typically larger and more aggressive than rainbow trout, and are voracious, piscivorous predators once they grow to a large size. They were a significant contributing factor of the extinction of McCloud River bull trout (Moyle 2002 p. 296). Brook trout were found to have an age specific biotic effect on cutthroat trout (Peterson et al. 2004).

Genetic introgression of non-native trout with native trout is another problem. Studies have been published on the adverse effects of hybridization between native and non-native trout including cases of hybrid vigor (Seiler et al. 2009) and hybrid swarm (Jug et al. 2004, Muhlfeld et al. 2009). Genetic introgression in higher elevations has been documented to have occurred in Kern River rainbow trout and Little Kern golden trout as a result of decades of fish stocking (Boiano et al. 2005, Erickson et al. 2010). In the Kaweah surveys, it was noted that there may be hybrid golden-rainbow trout present in the mid (1500m – 2500m) elevation sites (Werner 1981).

### ***Moving from Current Condition to Stewardship***

#### **Little Kern golden trout**

*Past Management.* Over the past 40 years, there has been extensive management on behalf of Little Kern golden trout. In accordance with the subspecies being listed under the ESA, CDFG, USFWS, SEKI, and Sequoia National Forest (SNF) enacted the “Little Kern Golden Trout Fishery Management Plan” in 1978 (revised in 1984). Under the plan, habitat restoration and fish removal were implemented throughout the Little Kern drainage. Restoration actions included the construction of fish barriers, chemical piscicide treatment of streams containing non-native trout, and restocking of Little Kern golden trout into treated areas. Potential source populations were analyzed using various genetic markers, particularly allozymes, and those that appeared to have no introgression with non-native rainbow trout were selected for use in restocking efforts. Recent genetic studies using both microsatellite and single nucleotide polymorphism (“SNP”) markers have shown these efforts to have been relatively successful. Many restored populations,



including those within SEKI, show little or no introgression. Other populations, however, still show genetic evidence of non-native rainbow trout (Stephens and May 2010). This could be due to incomplete eradication or the use of fish that were not pure Little Kern golden trout for restocking efforts. These introgressed individuals are found primarily in the middle and lower mainstem Little Kern River (Stephens and May 2010). A genetic management plan and a revised management plan are currently in progress for Little Kern golden trout.

*Critical Data Gaps.* While there has been extensive research on rainbow trout, there has been relatively little study of Little Kern golden trout, particularly its life history or population dynamics. Many characteristics are presumed to be similar to California golden trout, about which there has been more research, but have not actually been recorded for Little Kern golden trout. As such, natural levels of population connectivity, population size (and therefore natural levels of inbreeding), and specific habitat requirements have not been adequately documented. These characteristics, along with others such as temperate tolerance, could be important for future management.

Despite a lack of research aimed specifically at Little Kern golden trout, enough is known about its conspecifics (i.e. California golden trout) that general predictions about population response to the effects of climate change and other threats are likely to be fairly robust.

*Summary and Recommendations.* The outlook for Little Kern golden trout is mixed. Intensive management has led to the reintroduction of the fish into a large part of its historic range, though this work is not yet fully complete. The Little Kern golden trout within SEKI show little introgression, and are presumed to be at a stable population size (Danny Boiano, pers. comm.). However, there is only one known intact population within SEKI, and only four intact populations outside SEKI. These few populations are vulnerable to effects from fire, climate change, and potential future spread of non-native fish. Maintaining the protections currently in place for this population is the best strategy moving forward. Additionally, continued assistance to management efforts outside the park boundary, including development of a genetic management plan and revision of an overall management plan, would help ensure the complete recovery of this subspecies.

#### Kern River rainbow trout

*Past Management.* The CDFG, SNF, and SEKI created the Upper Kern Basin Fisheries Management Plan in 1995. Currently being revised, the plan outlines research needs for Kern River rainbow trout, particularly complete genetic estimates of introgression within the Kern River rainbow range, and the effects of trout habitat alteration by beaver. The necessary genetic work is currently underway at UC Davis. To date, little or no restoration actions have been conducted on Kern River rainbow trout. Restoration similar to that performed for Little Kern golden trout is largely infeasible, because the native range of Kern River rainbow trout is not located in headwater habitats. As such, restoring Kern River rainbow to their native range would have to include eradication of non-native and hybridized fish from essentially the entire Kern River watershed within SEKI. While technically not impossible, the probability of successfully eradicating fish from this extensive area is extremely low. Nevertheless, recent surveys appear to have detected relatively-pure Kern River rainbow populations in SEKI that were transplanted outside of their native range into headwater basins. Additional study of these populations has

been called for in order to assemble a more-complete understanding of the genetic integrity of these populations.

*Critical Data Gaps.* The data needs for Kern River rainbow trout are very similar to those of Little Kern golden trout, and include basic life history and population dynamics. However, the most critical need for management is an accurate assessment of introgression throughout the range. Additionally, estimates of gene flow within the habitat are vital for understanding how best to contain introgression and plan for possible restoration.

As mentioned for Little Kern golden trout, extensive research about rainbow trout conspecifics will likely be helpful in making predictions about Kern River rainbow trout's response to changing conditions. Generally high tolerance for a range of conditions are characteristic of *O. mykiss*, which may indicate such tolerance exists in Kern River rainbow trout. Nonetheless, drastic changes in the form of drought, fire, or flood would have negative consequences on the abilities of Kern River rainbow to persist.

*Summary and Recommendations.* Kern River rainbow trout is in very poor condition within its native range, in which it is likely that all populations have more than relatively low levels of introgression. Although genetic studies have shown that populations with relatively low levels of introgression exist outside of its native range, there are no reliable estimates of the size of these populations. The heavy stocking of non-native fish into the majority of the Kern River rainbow's historic range make restoration of the subspecies difficult if not impossible. In considering management strategies for the future, SEKI will have to weigh the benefit to this endemic trout against the cost, feasibility, and negative impacts of possible restoration actions. The relative success of the Little Kern golden trout presents hope for the continued persistence of the Kern River rainbow, although the Kern River rainbow inhabits a much larger area, and the stocking history is much more extensive and varied. Protection of existing populations having low levels of introgression, aided by natural barriers to gene flow in many sections of the habitat, is a possible solution in the short term. Even this action, however, would require the confirmation of healthy, self-sustaining populations of pure Kern River rainbow trout, for which more data are needed.

Introduced Trout in High Elevation (> 2500 m) Water Bodies:  
(Rainbow trout, California golden trout, brook trout and brown trout)

*Past Management.* As a national park, SEKI is charged with preserving biodiversity. The detrimental effects from the introduction of non-native trout have been known for some time, and the NPS has taken steps towards reducing these impacts. For example, stocking in national parks was phased out beginning in 1975, and stocking in SEKI was terminated in 1988. To reverse the impacts of previous stocking efforts, researchers and SEKI have conducted non-native fish eradication in several locations and this work is ongoing (Vredenburg 2004, NPS 2010). Non-native fish have been successfully eradicated from 12 water bodies in high elevation areas between the years of 1997 and 2010 resulting in the recovery of mountain yellow-legged frog populations in many of these lakes (NPS 2010). An additional 10 water bodies are currently in the process of having fish removed and five more water bodies are planned for fish removal to be initiated in 2012. The successful removal of fish and recovery of native species is a great success and lends support to continuing these efforts. SEKI is currently drafting an environmental impact statement (EIS) which is considering the eradication of non-native fish from up to 80 additional water bodies and associated streams.

*Critical Data Gaps.* Moving forward there are four biological data gaps and one social data gap that need attention in the near future. First, determine the effect of fish removal on the recovery of benthic invertebrate and zooplankton communities in alpine lakes. Second, very little is known about the effect of non-native trout on stream systems in high elevation areas within the parks and research should be done to determine if the impacts are similar to what has been found in lakes. However, a study is currently being conducted in SEKI to evaluate the recovery of stream invertebrate communities following removal of introduced trout. Third, all fish removal up to this point has been done with physical methods, but in some geographic areas this technique will not be a viable option. Research needs to be done to determine the efficacy of alternative techniques for fish removal. Fourth, the recovery process for native fauna in alpine lakes is not well understood and research into the degree of connectivity among populations of native species is needed to establish how management actions will need to occur for full recovery of alpine lake systems.

A social connection that is not well understood with regard to non-native fish removal is the effect it may have on recreational anglers and potentially local cities and towns that are somewhat economically dependent on tourism from anglers. However, after accounting for all past, current and potential future restoration, hundreds of waters will still contain fish and they are well distributed throughout SEKI's high elevations. It is unknown how or if restoration efforts will affect recreational anglers and local economies, and thus it would be beneficial to conduct research into the potential social and economic ramifications of fish removal. Currently, the parks have little information on how many people visit the high country for the stated purpose of fishing, but do have anecdotal observations from wilderness rangers and survey forms filled out by anglers visiting selected areas of the Kern, Kings and Kaweah Rivers. This potential conflict embedded in the NPS mission to conserve native biodiversity and provide for visitor use such as angling would benefit greatly from a systematic study of visitors and their preferences for fishing of non-natives within SEKI.

*Summary and Recommendations.* Due to the positive influence of non-native fish removal on mountain yellow-legged frog populations and potentially other native vertebrates and invertebrates, it is recommended that removal of fish continues from lakes where they are not native. Of the 573 water bodies in which non-native fish had become established, only 2% have had fish removed as of 2010, and an additional 15 lakes will likely have fish removed in the near future and bringing the total to 546 lakes still containing non-native fish. Furthermore, by ~10 years after 1988 (when stocking of 84 lakes was terminated in SEKI), , many fish populations went extinct (32%, Armstrong and Knapp 2004); and ~12 years have passed since this study was conducted in 1998-2000). It is therefore recommended that a survey of the water bodies reported to contain non-native trout by Armstrong and Knapp (2004) be conducted to gage if any further natural loss of trout populations occurred. The parks should evaluate both the biological and social impacts continued fish removal will have on the status of these systems, and if the above data gaps are addressed it will give future managers the ability to make informed decisions. Lastly, as climate change factors influence the system, it is recommended that a long term monitoring program for native aquatic fauna in high elevation water bodies be established, such that management actions under future climate scenarios can be informed by past population trend data.

#### Introduced Trout at low (< 2500m ) elevation (Rainbow trout, brook trout and brown trout)

*Past Management .* One management option that has been used in low elevation streams adjacent to SEKI to help keep non-native trout out of native trout waters is artificial fish barriers. Fish barriers were constructed in the Golden Trout Wilderness on the South Fork Kern River at Monache Meadows, as well as on the Little Kern River at Rifle Creek. Similarly, explosives were used to create a natural fish barrier at the mouth of Soda Spring Creek, a tributary of the Little Kern River in the Golden Trout Wilderness. All of these barriers were constructed to prevent upstream migration of non-native fish.

*Critical Data Gaps.* To better understand the impacts of non-native trout on the native trout found within SEKI, an increased understanding and quantification of specific interactions between these fish and their environment is needed. This would include studies of the age, size, and seasonality of both native and nonnative fish, and how each of these variables affects interactions such as predation and competition. In addition to the lack of population data available, presence-absence data could be a little more robust with respect to individual waters in SEKI. Presence of other non-native fishes such as green sunfish should also be documented if they are encountered, since they have already been found in the Kaweah River surveys.

*Summary and Recommendations.* Non-native salmonids are very well established within SEKI, and in national forest lands and designated wilderness areas adjacent to SEKI. Ecologically, SEKI should focus on eradication from critical habitat within park boundaries (and adjacent land where possible), primarily in isolated sections of streams inhabited by endemic fishes and historically fishless waters. Maintaining and installing artificial migration barriers should be considered to prevent natural reintroduction of nonnative fishes. Use of volunteer organizations can be helpful for providing assistance to restoration efforts. Additionally, studies of electrofishing as a means of non-native fish removal from some streams have shown a positive

result when completed over consecutive years (Carmona-Catot et al. 2010, NPS 2010). This may be a potentially useful strategy on lower flow stream segments where there is a series of fish barriers, with nonnative fishes residing between the barriers.

Monitoring efforts would be more informative if they were completed both more frequently and more broadly. While semi-decadal surveys such as the Kaweah transects are very useful, a lot of information is unavailable concerning annual variation in population dynamics and environmental interactions in the years between the surveys. Creating a few key representative sites to monitor on a more frequent basis would be an informative investment. Additionally, expanding the practice of utilizing voluntarily reported fisherman catch data such as presence-absence, length, and catch-effort would also help provide information at a broader spatial scale than is currently represented. This expanded information base would be extremely useful for considering potential changes due to a changing climate.

Lastly, non-native salmonids are not the only non-native predatory fishes present in low elevation stream reaches. Both green sunfish (*Lepomis cyanellus*) and black bullhead (*Ameiurus melas*) have been detected during surveys in the North Fork Kaweah River. In the case of the green sunfish, it represented one-third of the catch at the NFLN site in 2007. Though this report focused only on salmonids, any fish removal activities should take all non-native fish species into consideration. In general, long-term monitoring data reveal large decreases in rainbow trout where green sunfish are present. An additional consideration should be the decreased catch of native non-trout species such as Sacramento suckers (*Catostomus occidentalis*) when brown trout are present, as noted in the Kaweah River surveys at the NFLN site (NPS 2011).



## D. Invasive Species

### D.1. New Zealand Mudsnail *Potamopyrgus antipodarum*

#### **Abstract**

The New Zealand mudsnail is an introduced invasive native to New Zealand. It is very small and can quickly populate water bodies at very high densities. It has not been observed in the park, but has been seen just east of the park in Crowley Lake, Owens Lake, and Mono Lake drainages.

- **Species global status:** G5, Least Concern
- **Park status:** absent

#### **Species background**

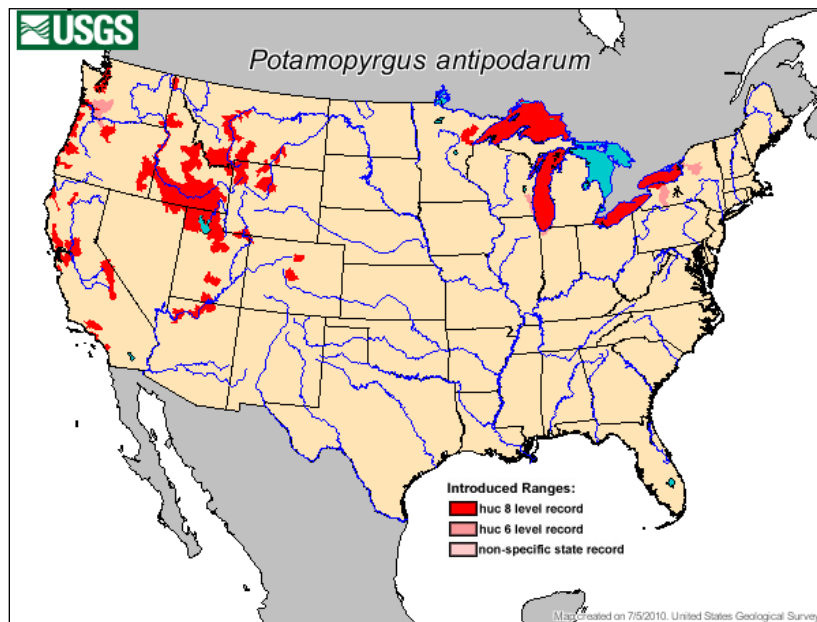
*Autecology, Habitat associations, life-history, taxonomy.* *Potamopyrgus antipodarum* (Gray 1843) (Gastropoda, Hydrobiidae) is a small, nocturnal grazer that has high physiological tolerances. It prefers to eat ephiphitic and periphytic algae, sediments, or diatoms. This snail can be found in a wide variety of aquatic habitats including flowing or still waters, fresh or estuarine habitats, and rocky or littoral substrates (Winterbourn 1970). *Potamopyrgus antipodarum* can grow and reproduce in habitats with heavy siltation, with salinity up to 26 parts per thousand, in temperatures between 0 and 34°C, and survive short periods of desiccation or salinities greater than 30 parts per thousand (Winterbourn 1970, Cox and Rutherford 2000). It thrives in disturbed watersheds and areas with high nutrient loads (Zaranko et al 1997). One study showed that the snails did not do well in streams with specific conductivity below 200 µS/cm in the Owens River (Herbst et al. 2008). It is possible that water bodies in the parks are less susceptible to invasion by *P. antipodarum* than other water bodies because most park waters have relatively low sediment and nutrient loads, are relatively undisturbed, and have low conductivities.

*Potamopyrgus antipodarum* is ovoviviparous (young are live born) and parthogenetic (embryos may mature without male fertilization producing haploid offspring). All of the New Zealand mudsnails in the Western US belong to one of two clones, or morphs, but it is not yet known if these different morphs display different life history characteristics. Over 95% of these snails are female, and in the course of its annual lifecycle one female can produce an estimated 230 offspring (Lassen 1979). These life-history traits contribute to the highly invasive nature of *P. antipodarum*.

#### **Regional Context**

*Distribution in the lower 48 states.* *Potamopyrgus antipodarum* is found in nine western US States, all Great Lakes, with possible occurrences in two eastern states outside the Great Lakes (Figure D1.1). The first detection of *P. antipodarum* in North America was in the Middle Snake River, Idaho, in 1987 where it is likely to have been introduced through the transport of live gamefish (Zaranko et al 1997). Since this introduction the snail has spread rapidly throughout the West (Figure D1.2). By 1989 it had expanded to become the most abundant mollusk in the Middle Snake River (Bowler 1991). By 1997 its range covered approximately 640 km in the Snake River as well as many of its tributaries, and had moved across the continental divide into the Madison River of the Missouri River system. In addition to this localized dispersion, the snail began to appear in watersheds throughout the west, from southern California to Puget Sound

(USGS NAS program), likely due to unintentional movement by humans. The snail was first detected in the Great Lakes in 1991 (Zaranko et al 1997), and is suspected to have been introduced separately through ballast water. Due to their small size, clonal reproduction, high tolerance for desiccation, and incredible density when established, these snails are prime candidates for successful transport and invasion of new habitats.

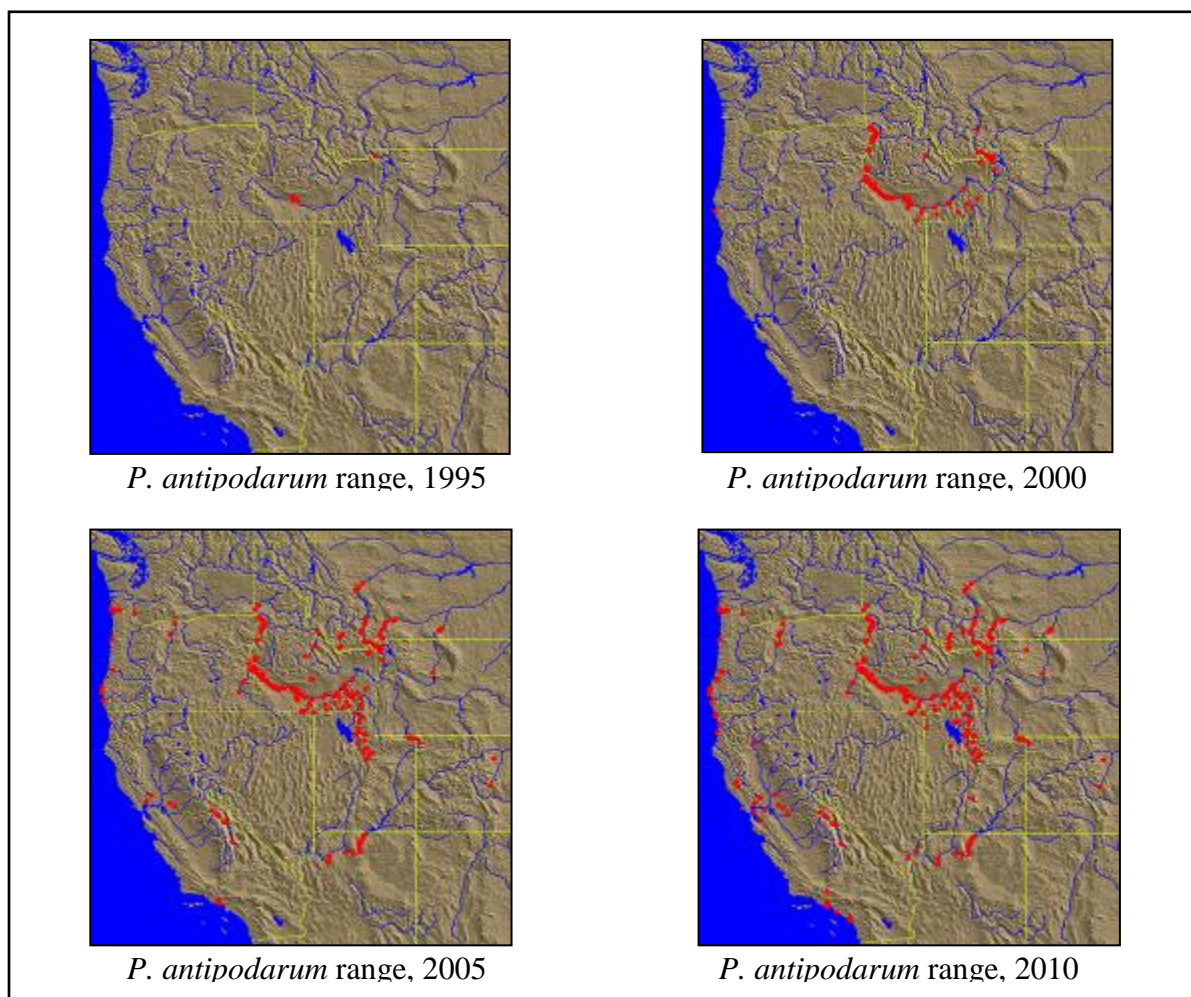


**Figure D1.1.** Current United States distribution of *Potamopyrgus antipodarum*. Data compiled by the Nonindigenous Aquatic Species Program, USGS.

### ***Distribution in Sequoia and Kings Canyon National Parks***

*Potamopyrgus antipodarum* has not been found in Sequoia or Kings Canyon National Parks. However, the species is documented in the Crowley Lake, Owens Lake, and Mono Lake drainages, located just east of the parks (Figure D1.3, Table D1.1). Although documented primarily in regional lakes, the species is also present in western streams and the concern is that the species could move upstream and into park locations.

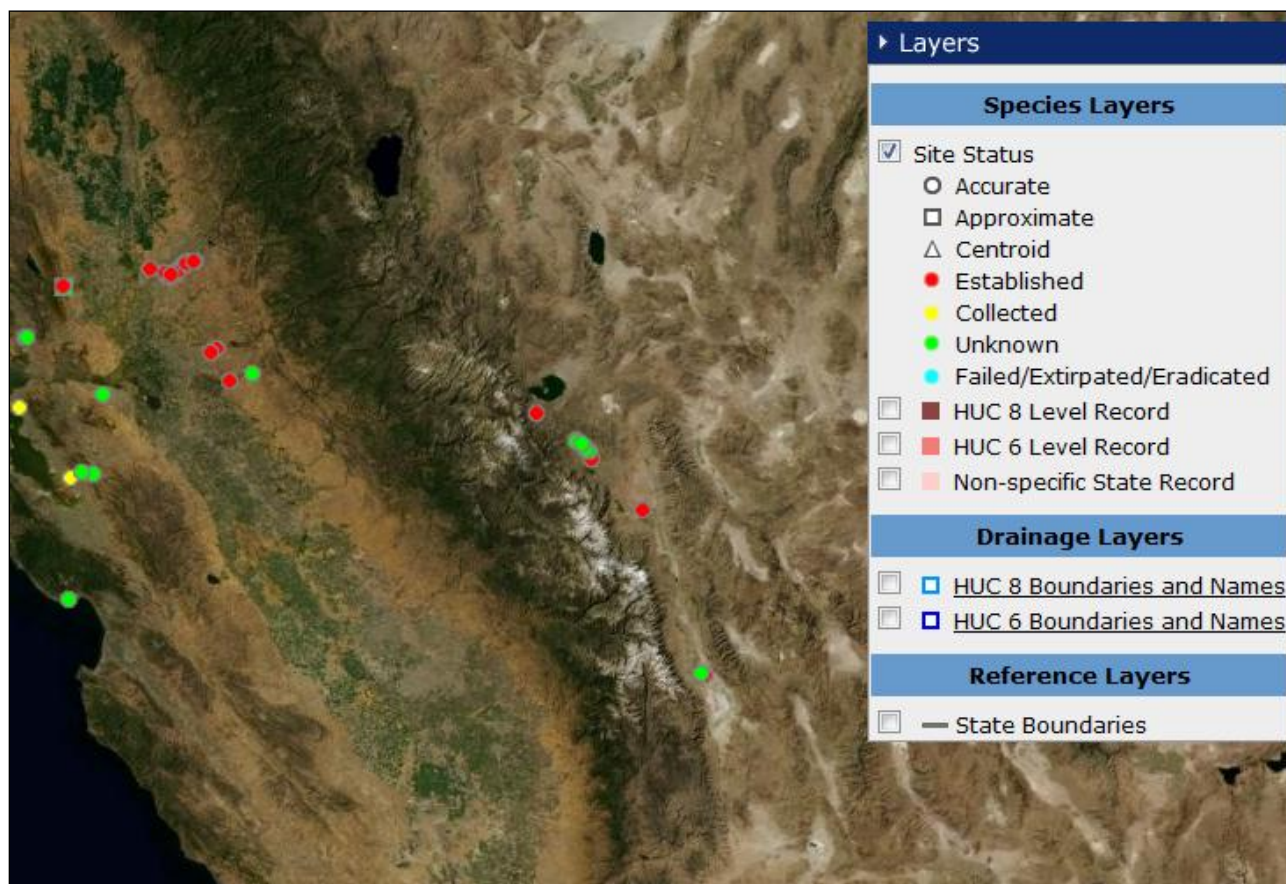




**Figure D1.2.** Maps showing the invasion of *Potamopyrgus antipodarum* in the Western US. Data are from <http://esg.montana.edu>.

### ***History of spread, introduction mechanisms***

The main factor in whether or not a freshwater area is invaded by *P. antipodarum* is simply if the species has been introduced there. In many environments it quickly takes over once it is introduced, and it is predicted by models to be able to spread quickly in most parts of North America if introduction into an area is not prevented (Loo et al. 2007). *P. antipodarum* can be spread through water bodies such as connected streams and rivers. This is not a high risk for introduction into the parks because many water bodies in the parks begin within the parks. However, if the snail is introduced to any water body within the parks, connected water bodies will be at high risk for invasion. It is also possible for *P. antipodarum* to move slowly upstream, since it can easily be introduced to protected areas through the boots of hikers and the equipment of managers and researchers. Therefore, anyone working in currently invaded areas should be especially diligent in cleaning their equipment. In addition, these snails may be transported by animals, including waterfowl (Larsen 1978) and cattle (New Zealand Mudsail Management and Control Plan Working Group, 2007).



**Figure D1.3.** Point locations of *P. antipodarum* in region surrounding Sequoia and Kings Canyon National Parks. Background map courtesy of NASA, data from USGS Nonindigenous Aquatic Species Program, <http://nas.er.usgs.gov>.

*Spatial structure, connectivity issues.* *P. antipodarum* can spread through contiguous water bodies such as connected streams and rivers. It easily and quickly disperses downstream, and is also capable of slow upstream movements. Zaranko et al (1997) recorded the snail spreading 60 m upstream in three months' time. Because the headwaters of all stream systems in Sequoia and Kings Canyon National Parks begin within the park, there is less risk of invasion from currently established populations at lower elevations, but managers still need to be vigilant in monitoring the spread upstream from known *P. antipodarum* populations. If the snail is introduced to any water body within the parks, all connected lakes and streams will be at high risk for invasion.

*Habitat associations, biological and geological specificities.* *P. antipodarum* is able to live in a wide variety of aquatic habitats including flowing or still waters, heavy siltation, surface water freezing, and periods of desiccation (Alonso & Castro-Diez 2008). It thrives in disturbed watersheds and areas with high nutrient loads (Schreiber et al. 2003). It is possible that water bodies in the parks are less susceptible to invasion by *P. antipodarum* than other water bodies because most park waters have relatively low sediment and nutrient loads, are relatively undisturbed, and have low conductivities.

Table D1.1. *Potamopyrgus antipodarum* observations near Sequoia and Kings Canyon National Parks. Data are from the USGS Nonindigenous Aquatic Species information resource; <http://nas.er.usgs.gov>.

State	County	Locality	Year	Drainage	Status	Spec. ID#
CA	Mono	Owens River between the mouth of Hot Creek and Benton Crossing [~3 miles N of Lake Crowley]	2001	Crowley Lake	collected	<u>52740</u>
CA	Mono	Benton Crossing	2002	Crowley Lake	established	<u>157442</u>
CA	Inyo	Bishops Creek canal	2002	Crowley Lake	established	<u>157443</u>
CA	Mono	Owens River at mouth of Crowley Reservoir	2001	Crowley Lake	established	<u>157444</u>
CA	Inyo	Lone Pine Creek [Lone Pine, CA]	2004	Owens Lake	established	<u>165692</u>
CA	Mono	Rush Creek	2004	Mono Lake	established	<u>242907</u>
CA	Mono	Rush Creek [at Hwy 395 crossing, near intersection of SR 158, ~5 mi S of Mono Lake]	2008	Mono Lake	unknown	<u>256609</u>
CA	Mono	Owens River [near mouth of McLaughlin Creek]	2008	Crowley Lake	unknown	<u>256610</u>
CA	Mono	Owens River [near mouth of McLaughlin Creek]	2008	Crowley Lake	unknown	<u>256611</u>
CA	Mono	Owens River [~1 mi E of McLaughlin Creek mouth]	2008	Crowley Lake	unknown	<u>256612</u>
CA	Mono	Owens River [~2 mi SE of McLaughlin Creek mouth]	2008	Crowley Lake	unknown	<u>256613</u>
CA	Mono	Owens River [~2 mi SE of McLaughlin Creek mouth]	2008	Crowley Lake	unknown	<u>256614</u>
CA	Mono	Owens River [~2.5 mi SE of McLaughlin River mouth]	2008	Crowley Lake	unknown	<u>256615</u>
CA	Mono	Owens River [~2.5 mi SE of McLaughlin Creek mouth]	2008	Crowley Lake	unknown	<u>256616</u>
CA	Mono	Owens River [~2.5 mi SE of McLaughlin Creek mouth]	2008	Crowley Lake	unknown	<u>256617</u>
CA	Mono	Hot Creek [near confluence with Owens River]	2008	Crowley Lake	unknown	<u>256618</u>
CA	Mono	Owens River [near mouth of Hot Creek]	2008	Crowley Lake	unknown	<u>256619</u>
CA	Mono	Owens River [near mouth of Hot Creek]	2008	Crowley Lake	unknown	<u>256620</u>
CA	Mono	Owens River [near mouth of Hot Creek]	2008	Crowley Lake	unknown	<u>256621</u>
CA	Mono	Owens River [at Benton Crossing Rd.]	2008	Crowley Lake	unknown	<u>256622</u>
CA	Mono	Owens River [at Benton Crossing Rd.]	2008	Crowley Lake	unknown	<u>256623</u>
CA	Mono	Owens River [~0.5 mi N of Lake Crowley]	2008	Crowley Lake	unknown	<u>256624</u>
CA	Mono	Owens River [~0.25 mi N of Lake Crowley]	2008	Crowley Lake	unknown	<u>256625</u>
CA	Inyo	Los Angeles Aqueduct [in Lone Pine]	2008	Owens Lake	unknown	<u>256626</u>

### **Description of threats from *P. antipodarum***

The biotic and abiotic effects of introduced *P. antipodarum* on aquatic ecosystems has varied greatly among studies and is likely highly dependent on the specific aquatic community being invaded. These snails have been shown to be substantial grazers, eating as much as 75% of gross primary production in studied streams in Wyoming (Hall et al. 2003). They were shown to affect



ecosystem function by dominating nitrogen and carbon cycling in one of these streams, and particularly by increasing nitrogen fixation (Hall et al. 2003, Hall et al. 2006, Arango et al. 2009). Many fish species, including brown trout, were shown not to preferentially feed on these snails (Cada 2004). They competed with native aquatic invertebrates in the Greater Yellowstone ecosystem (Kerans et al. 2005). These snails have been shown to outcompete native snails and cause native snail populations to decline in Poland (Strzelec 2005). *P. antipodarum* was shown to outcompete native invertebrates and constitute up to 92% of the invertebrate production in a stream in Wyoming (Hall et al. 2006). However, they were found to have no detectable effect on other invertebrates or on primary productivity in the Colorado River (Cross et al. 2010).

### **Management**

Prevention of introduction is the best management option for *P. antipodarum*. It can be introduced on boots and equipment of visitors and staff. Care should be taken to thoroughly clean all boots and equipment that have been used outside of the park, especially in areas which contain the snail. Information for visitors concerning *P. antipodarum* and how to clean their boots could be posted in visitor centers and park websites. Monitoring of water bodies for the snail can aid in early detection of the snail in the parks.

*Description of role of Sequoia/Kings Canyon in the containment of impacts of P. antipodarum.* Because the headwaters of all stream systems within Sequoia and Kings Canyon National Parks begin within the parks, the primary role in the containment of *P. antipodarum* lies in preventing its establishment within the parks, and thus reducing the likelihood of its spread to the rest of the Southern Sierra and Central Valley. Five major rivers originate within SEKI (Kings, Kaweah, SJ, Kern and Tule), and if *P. antipodarum* were to establish in these watersheds, the entire San Joaquin valley could be colonized.

### **Summary and Management Recommendations**

The New Zealand mudsnail is a invasive, nonnative that is not currently present in the parks, but could cause substantial ecological impacts if it becomes established. Prevention of introduction is the best management option for *P. antipodarum*. It can be introduced on boots and other equipment of visitors and staff. Care should be taken to thoroughly clean all boots and equipment that have been used outside of the park, especially in areas which contain the snail. Information concerning the impacts of *P. antipodarum* could be posted in visitor centers and park websites to educate visitors about the risk. A program dedicated to educating visitors and park staff about proper cleaning procedures for boots and equipment, as well as stations for such cleaning, could also help reduce the risk of introduction. Monitoring of water bodies for the snail can aid in early detection if this species becomes established in the parks.

If the snail is detected in the parks, rapid eradication is suggested before the invasion spreads. Eradication methods include periodic molluscicide or biocide application, periodic desiccation of waterbodies, periodic introduction of biological control agents, and/or mechanical methods. These methods are detailed in the National Management and Control Plan for the New Zealand Mudsnail, pages 27-31, as produced by the New Zealand Mudsnail Management and Control Plan Working Group for the Aquatic Nuisance Species Task Force, which can be found at [http://www.anstaskforce.gov/Documents/NZMS\\_MgmtControl\\_Final.pdf](http://www.anstaskforce.gov/Documents/NZMS_MgmtControl_Final.pdf).

## D.2. Brown-headed Cowbird (*Molothrus ater*)

### **Abstract**

Brown-headed Cowbirds are nest parasites who can exacerbate problems for their host species caused by habitat loss and degradation. Host species most vulnerable to cowbird parasitism are likely to be neotropical migrants with a short breeding season and long incubation period, or species with small geographic ranges. Cowbird populations can be discouraged by establishing and maintaining contiguous forest cover and by avoiding the creation of cowbird food sources. In some cases, trapping may be necessary to protect local populations of sensitive species.

- **Degree of invasion:** Significant (fairly common summer resident and regular breeder)
- **Degree of invasiveness:** High (rapid range expansion in response to habitat change)
- **Impacts in invaded habitats:** Significant (may parasitize nests of over 200 species)
- **Control status / recommendations:** Careful monitoring of cowbird populations and their impacts on songbird reproductive success is recommended, along with efforts to minimize and concentrate cowbird food sources.

### **Species Background**

The Brown-headed Cowbird (*Molothrus ater*) is a widespread songbird native to North America that is infamous for its reproductive strategy of nest parasitism. It is a member of the Icteridae family in the order Passeriformes, with two subspecies that breed in California: Sagebrush Cowbird (*M. a. artemisiae*), a rare summer resident in the Great Basins region east of the Sierra Nevada, and Dwarf Cowbird (*M. a. obscurus*), which is much more common. Differences in morphology and vocalizations have been supported by genetic work, though there is substantial gene flow in the Sierra Nevada (Fleischer et al. 1991).

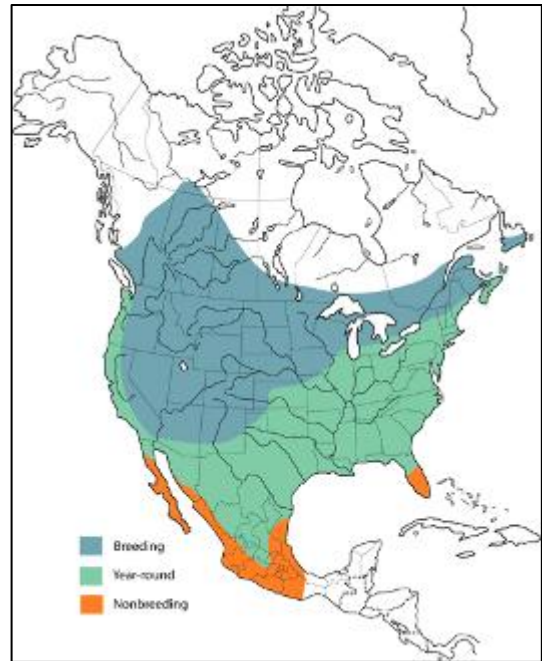
A short-distance migrant, females usually arrive on their breeding grounds from mid-April to early May and use high perches to begin searching for host nests in which to lay their eggs (Uyehara et al. 2000). In much of their range, egg-laying can begin in late April at a rate of 0.5 to 0.8 eggs per day and continue through July, depending on the specific location, weather conditions, and the age of the female (Uyehara et al. 2000). A female cowbird can lay 30-40 eggs in a single season, and can therefore parasitize many nests. Over 200 species of hosts, especially songbirds, have been documented.

### **Distribution and population trends**

*Global and regional context.* The Brown-headed Cowbird currently occupies most of North America, south of the Arctic (Figure D2.1). They are commonly associated with livestock, and attracted to foraging areas with short grass and high invertebrate densities or grain seeds, particularly anthropogenic food sources, including: grain crops, feed lots, pastures, horse corrals, golf courses, bird feeders, and camp grounds (Uyehara et al. 2000). In winter, cowbirds can reach high densities as they concentrate in rice fields, dairies, and feed lots (Laymon 1987). Breeding grounds are typically a mosaic of open and wooded habitat within 10km of a food source.

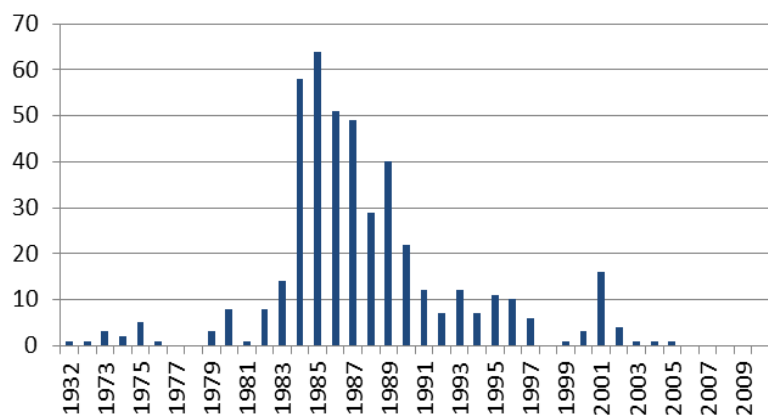
Historically, cowbirds were probably limited to open grasslands mostly west of the Mississippi River. They were associated with migratory bison herds and foraged in short grass and on bare ground for insects and prairie grass seed (Robinson et al. 1993). Over the last 200 years, however, they have undergone a rapid range expansion, colonizing the eastern one-third of the United States by the early 1800s, the Canadian Maritime provinces by the 1900s, and the southeastern United States by the late 1950s and early 1960s (Rothstein and Peer 2005). There are no records west of the Colorado River before 1870, and the population expanded rapidly west and north between 1900 and 1930 (Laymon 1987).

This range expansion and population increases have been attributed to many anthropogenic factors, including: the fragmentation of continuous forest to a mosaic of forests, fields, pastures, roads and towns; increased winter food supply in the form of waste grain in agricultural fields; and increased reproductive rates as they encountered new host species with little to no defenses against nest parasitism. Although there are still small areas of local increase and range expansion, Brown-headed Cowbirds are currently experiencing an overall decline in population size. Breeding Bird Survey data indicates a survey-wide decline of 1.1% (95% CI: 0.8-1.3%) over the period 1966-2007 (Sauer et al. 2008). In the Sierra Nevada of California, the decline is a steeper 3.9% (95% CI: 2.2-5.5%) over the same period (Sauer et al. 2008), while populations in the Central Valley of California may actually be increasing (Rothstein and Peer 2005).

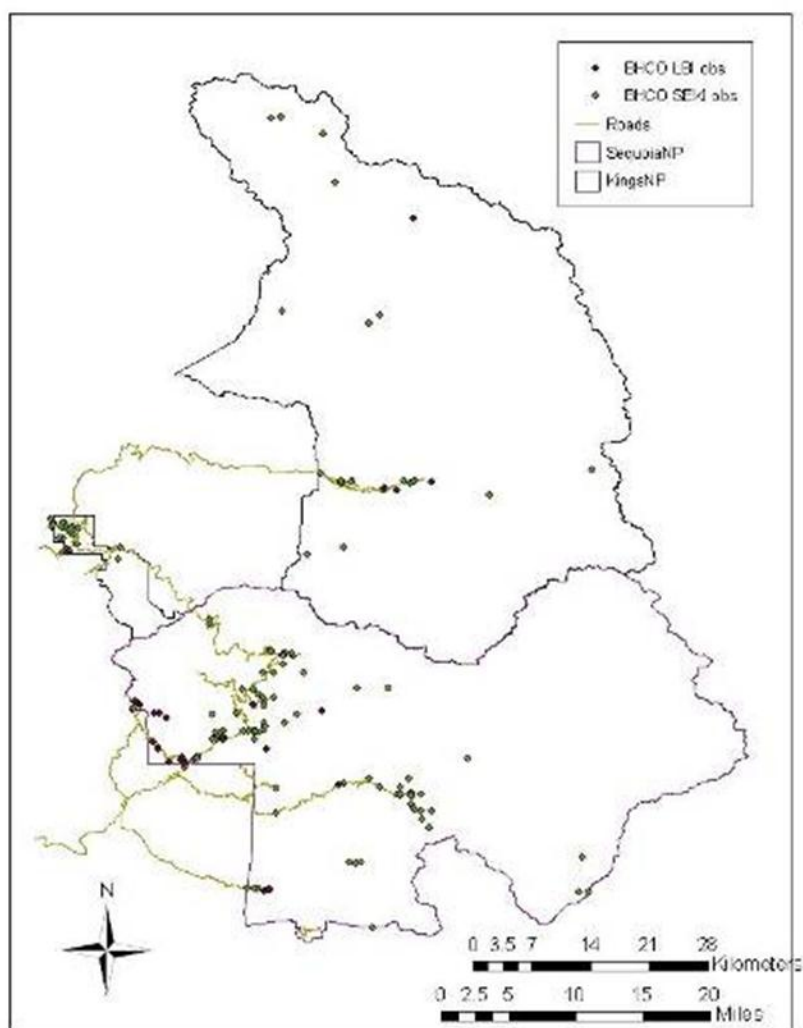


**Figure D2.1.** Brown-headed Cowbird distribution (Lowther 1993)

*Sequoia-Kings Canyon distribution and population trends.* Within the park, recorded observations of Brown-headed Cowbirds peaked in the 1980s, with few to no observations recorded in recent years (Figure D2.2). The majority of these recorded observations are in lower elevation sites and near roads, although Brown-Headed Cowbirds have been observed throughout much of the park (Figure D2.3). Similarly, in a recent landbird inventory of Sequoia/Kings Canyon, the majority of Brown-Headed Cowbird detections were in locations classified as Blue Oak Woodland, California Black Oak Forest, Mixed Chaparral, Ponderosa



**Figure D2.2.** Recorded observations of Brown-headed Cowbirds in Sequoia and Kings Canyon (SEKI 2010).



**Figure D2.3:** Map of Brown-headed Cowbird observations in Sequoia and Kings Canyon (SEKI 2010)

Pine Woodland and Canyon Live Oak Forest habitats (Table 1; Siegel and Wilkerson 2005). Habitat types dominated by oak trees, including Blue Oak Woodland and Interior Live Oak-Canyon Live Oak-California Buckeye Woodland, had the highest estimated density of Brown-Headed Cowbirds (Table D2.1), suggesting that host species in these habitat types may have the highest potential for parasitism. However, it is important to note that cowbird counts may not correlate well with nest parasitism rates. Many males may be unmated, so future counts should distinguish between observations of males and females. Female counts may be a better indicator of parasitism rates, though rates are likely to vary quite a bit from host species to host species (Robinson et al. 1993).

**Table D2.1.** Brown-headed Cowbird detections and estimated densities by habitat type. Data are from a landbird survey conducted by the Institute for Bird Populations in 2003-04 (Siegel and Wilkerson 2005). Density estimates are adjusted for detectability of the species.

Habitat Classification	% of plots with detections	Estimated density (birds/ha)
Blue Oak Woodland	36.8%	0.49
CA Black Oak Forest	33.3%	0.03
Interior Live Oak-Canyon Live Oak-CA Buckeye Woodland	14.8%	0.15
Mixed Chaparral	8.9%	0.07
Ponderosa Pine Woodland	6.3%	0.06
Canyon Live Oak	6.3%	0.06
Ponderosa Pine – Incense-cedar Forest	1.9%	0.02
Giant Sequoia Forest	1.3%	0.02
Montane Chaparral	1.3%	0.01
White Fir/Sugar Pine Forest	1.0%	0.01
Red Fir Forest	0.7%	0.01

### **Conservation concern**

Brown-headed Cowbirds have been targeted as a contributing factor to the range-wide decline of many songbird populations, especially neotropical migrants. It is clear that they can have a negative impact on the host's nest success rates, including complete destruction of a host's clutch or brood to force re-nesting, removal of some host eggs, increased likelihood of nest abandonment, and diversion of food resources from host nestlings to larger, louder cowbird nestlings (Smith et al. 2002; Robinson et al. 1993). However, because many birds can re-nest, or are successful in producing young despite parasitism, there may actually be little net loss in host reproductive success over the entire breeding season, and it is less clear how important parasitism is on a population level (Smith et al. 2002). Even in host populations with near 100% nest parasitism rates, reproductive success may be more impacted by high nest predation rates (Rothstein and Peer 2005). Instead, high nest parasitism and nest predation rates can be seen as side effects of habitat degradation due to agriculture, grazing, and development.

Host species most vulnerable to population-level effects of cowbird parasitism are likely to be neotropical migrants with relatively short breeding seasons and long incubation periods, and hosts with small geographic ranges (Robinson et al. 1993; Rothstein and Peer 2005). To determine the extent and impact of cowbird parasitism on the bird communities of Sequoia/Kings Canyon National Park will require a comprehensive nest monitoring study. The



critical parameters to measure in order to determine the effect on a host population are: (1) parasitism frequency, (2) nest predation frequency, (3) frequency of abandonment of parasitized and unparasitized nests, (4) the number of host young fledged from parasitized and unparasitized nests that escape predation, (5) the length of the nest cycle, (6) the length of the incubation period, and (7) the length of the breeding season (Robinson et al. 1993).

### ***Management***

Cowbird populations have been managed through lethal control, including trapping and killing adults, adding or removal of cowbird eggs from host nests, and shooting cowbirds at roost sites. Cowbird trapping can be an effective way to remove a large number of cowbirds on a local scale. Trapping also requires continuous and sustained effort to be effective, which may be cost prohibitive on a large scale. Resistance to trapping and ability to escape from traps appears to be spreading through cowbird populations, and killing can produce a public backlash, so active cowbird control should only be used where it is absolutely essential (Rothstein and Peer 2005). However, even in the case of rare and endangered species, a better use of resources may be to increase available habitat and invest in improvements to habitat quality, such as reforestation to reduce forest fragmentation and increase the width of riparian areas. Reforestation is a more long-term solution, requiring no continuing funding or person-power, benefitting the entire bird community, and may have the added benefit of reducing nest predation rates as well (Laymon 1987; Rothstein and Peer 2005). Another habitat management approach is to reduce, cluster, or eliminate cowbird feeding opportunities, such as stables, pack stations, bird feeders, or grazing areas (Verner and Ritter 1983; Robinson et al. 1993). A radio-telemetry study in the Sierra Nevada documented a commute of up to 6.7km between breeding and feeding locations, and estimated that a single horse corral enabled cowbirds to parasitize an area of 154km<sup>2</sup> that otherwise contained no suitable feeding sites (Rothstein et al. 1984).

### D.3. Feral Pigs (*Sus scrofa*)

#### **Abstract**

Feral pigs are invasive ungulates that are native to Eurasia and North Africa and have been introduced globally. Feral pig populations have increased in California in recent years, leading to extensive ecosystem damage. Feral pigs are primarily controlled through use of exclusionary fencing and hunting.

- **Degree of invasion:** Minimal
- **Degree of invasiveness:** High
- **Impacts in invaded habitats:** Significant
- **Control status / recommendations:** Feral pigs are present during spring months along the southeastern edge of SEKI, but populations in SEKI are likely kept under control by mountain lions. At current levels, feral pigs pose limited threat to park resources, but continued monitoring is recommended.

#### **Background and taxonomy**

The feral pig (*Sus scrofa*) is a medium-sized ungulate that is native to Eurasia and Northern Africa and has been introduced to every continent except for Antarctica (Graves 1994). The feral pig is in the order *Artiodactyla*, family *Suidae*, and genus *Sus*. Feral pig populations in California are composed hybrids of feral domesticated pigs (*Sus scrofa domesticatus*), introduced by Spanish settlers in the 1700s, and European wild boars (*Sus scrofa scrofa*), introduced by a Monterey landowner for hunting purposes (Waithman 2001). Wild boars are characterized by long, abundant hair covering their bodies, small, erect ears, a straight tail tasseled at the end with long, coarse hairs, a lean, razor-backed body with shoulders that are higher and wider than the hindquarters, long sharp tusks and a long snout. Wild boars are predominantly black, pied or russet in color with offspring that are dark with horizontal stripes. In contrast, domestic pigs have short, sparse hair, large, floppy ears, curly tails, a uniformly wide body and flat back, short tusks and a short snout. Domestic pigs and their offspring are generally uniformly white, pink or russet in color. The appearance of most wild pigs in California falls somewhere in-between these two phenotypes depending on ancestry (Waithman 2001).

In 1957, wild pigs were designated a game mammal in the state of California and hunting is actively managed by the California Department of Fish and Game (Waithman 2001). Hunters must currently obtain a hunting license in order to kill a wild pig.

#### **Life History**

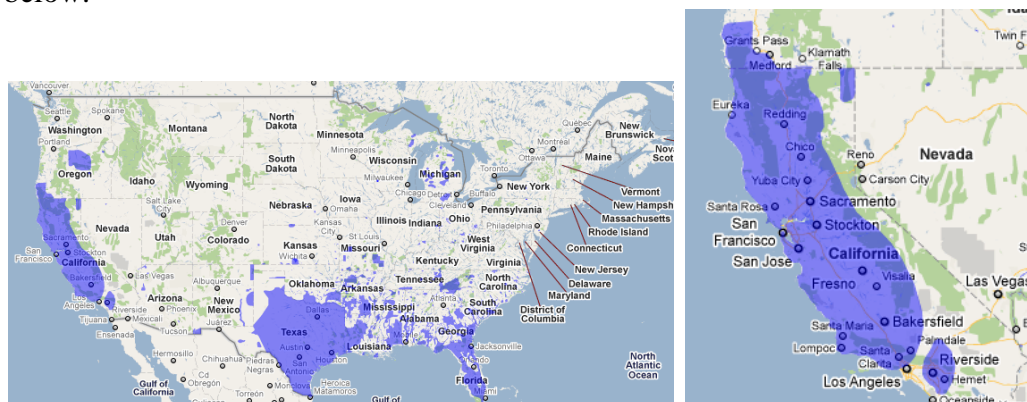
Feral pigs are found in a variety of habitats, including densely forested mountains, brushland, swamps, dry ridges, and meadows (NatureServe 2009). In California, feral pigs are most closely associated with oak woodlands, oak woodlands and mixed-conifer forests, oak grasslands, and chaparral shrublands (Mayer and Laudenslayer 1988). Feral pig populations in oak woodland habitat have been increasing in recent years and regional differences in density may be associated with acorn mast and access to permanent water sources (Schauss et al. 1990). Seasonal changes in habitat use are typically link to availability of food and home ranges tend to be smaller when food resources are available (Singer 1981). Feral pigs invade new habitat by dispersing in groups or as individuals (Singer 1981).

Feral pigs are omnivores and feed opportunistically on a variety of plant and animal sources. While they typically prefer to eat vegetation to animal matter, feral pigs will also eat invertebrates and vertebrates, both as carrion and live prey (Wilcox 2009). When foraging, pigs utilize rooting behavior, in which they tear up the ground with their snouts looking for roots and invertebrates. This behavior is highly disruptive to the environment and can destroy native vegetation and habitat for native animals.

In good environmental conditions, wild boars have high reproductive rates relative to other ungulate species. Females can reproduce up to 2 times per year with litters of 5-6 offspring (Waithman et al. 1999). Females typically reach sexual maturity at one year and as early as 6-8 months in great food abundance (Singer 1981). Mating typically takes place in the fall and spring, with offspring born in the following summer or winter. Reproductive rates are closely tied to food availability. In particular, mast events, such as acorn mast in oak woodlands, can result in large increases in population growth (Bieber and Ruf 2005).

### ***Distribution and population trends***

**Global and regional context.** *Sus scrofa* is present throughout parts of the western, southern, and southeastern United States (Figure D3.1), northwestern Australia, and many Indo-Pacific islands (Graves 1994). Globally this species is abundant in its native and introduced ranges. The wild boar from which the common domestic pig was domesticated is present throughout much of Eurasia and northern Africa (Graves 1994). This species has been growing in population in the United States since its introduction in the late 1800s. The current population estimates are 3-4 million feral pigs in the United States. Feral pigs in California have expanded their range and continued to grow in numbers over recent years, leading to populations ranging from 500,000-1,000,000 in California alone. Once established, feral pigs are difficult to eradicate, as discussed below.



**Figure D3.1** Map of the current distribution of feral pigs in the United States, and specifically in California. This map was created by the National Feral Swine Mapping System and is current as of November 7, 2010.

**Sequoia-Kings Canyon distribution and population trends.** Feral pigs have been observed along the southwestern boundary of Sequoia National Park. They tend to be present in the spring and move out of the park in the summer, possibly due to the availability of food resources in the park. This has been the case for many decades, and they do not seem to be an increasing threat at this time. It is thought that the healthy mountain lion population in the park is keeping the feral pig population under control (H. Werner, personal communication).

### **Conservation concern**

Feral pigs rank high among the most damaging nonnative species introduced to national parks and reserves in United States (Vitousek et al 1997). They inhabit many areas in the national park system, including the southeastern US, Hawaii and California (Singer 1981). They can have devastating effects on ecosystems, due to their rooting behavior. Rooting overturns native vegetation, damages the habitat of native animals that live on or under the ground, and loosens soil, which can cause erosion and reduce water quality. Studies have found that impacts of feral pigs can include: reduced plant cover, reduced macroinvertebrates, reduced voles and shrews, increased soil nitrates, increase soil ammonia and ammonium, decreased soil calcium, decreased soil magnesium, decreased soil cation capacity, decreased bulk density of soil, and increased leeching (Singer et al. 1982). Feral pigs also act as dispersers of invasive plants. In California oak woodlands, feral pigs negatively impact other species by competing with native animals for acorn mast crops and reducing oak recruitment through acorn consumption and seedling destruction.

### **Management**

Pig eradication programs have typically used a combination of hunting and exclusionary fencing. Feral pigs were successfully eradicated from Santa Cruz Island by fragmenting their population through fencing and hunting them by helicopter (Ramsey 2009). Pinnacles National Monument used a similar approach to eradicate feral pigs. The Monument constructed 24 miles of pig-proof fence around the perimeter of some of the park (McCann and Garcelon 2008). The Institute for Wildlife Studies was contracted to eradicate pigs within the boundaries using ground-hunting methods, trained dogs and traps. An eradication program such as this is expensive and requires ongoing management. Fences deep enough to prevent pigs from burrowing under them are expensive, require intensive upkeep, and would likely act as barriers to movement for other animals, so they should be used as a secondary measure, if sport hunting and natural predators are not sufficient to control populations.

Sport hunting in the state of California is managed by the Department of Fish and Game (Waithman 2001) and is a highly effective tool for reducing feral pig densities (Sweitzer et al. 2000). Sport hunting by individual sportsmen is not currently permitted within SEKI and thus would not be an appropriate control measure. However, feral pig hunting is allowed on other federal lands (including National Monument and National Forest) and private lands, so licensed hunters and the CA DFG could target source populations outside of the park. Management through hunting is unlikely to result in complete eradication of feral pigs, but can be used to reduce densities to acceptable levels.

Currently, feral pigs are present at the southwest border of the park, but do not present a problem requiring direct management. Monitoring the park for feral pigs, especially the southwestern area, should allow for early detection and management of any future problems. At present, *S. scrofa* is not established in SEKI. Therefore, the role of the parks in managing this species is to monitor for its encroachment into the park and attempt to control populations within the park if they do become established.

## D.4. Cattle Trespass

### **Abstract**

Cattle trespass has been a long-standing impact on natural areas of the Southern Sierra Nevada. Impacts can be divided into those that impact terrestrial vegetation, aquatic communities, wildlife impacts, and impacts that foster invasive species. Within SEKI, most cattle trespass impacts are highly localized around 8 locations some of which are historic and not on-going. There are 12 grazing allotments that abut the park. These are the areas where any future trespass is likely to occur. More needs to be done to monitor the long term impacts of these trespass events.

### Overall Impacts

- Shifts in plant community composition, including increasing prevalence of invasive species
- Degradation of water quality
- Changes in landscape hydrology due to soil compaction and stream bank erosion
- Negative impacts to some wildlife species
- Degradation of rare or sensitive ecosystem types (blue oak woodland and alpine meadow)

### Level of Threat

The presence of grazing allotments along much of the western boundary of the park makes the threat of cattle trespass substantial. While the level of damage caused by browsing cattle is highly variable, most trespass sites are in the southwest region of the park, where oak woodland dominates. As protected areas of this ecotype are rare, any threat to their integrity should be considered as serious.

- ***Degree of invasion:*** Minimal
- ***Degree of invasiveness:*** Moderate
- ***Impacts in invaded habitats:*** Localized but can be significant
- ***Control status / recommendations:*** Proactive engagement of neighboring ranchers to help coordinate a plan to minimize risk.

### **Background**

Systematic grazing of sheep and cattle in the Sierra Nevada has resulted in a variety of impacts on natural systems. Domestic grazing in the Sierras began in the 1860s. After an initial period of unchecked intensive grazing, state and federal lands were designated and grazing was regulated around 1900, largely due to rapid and widespread impacts to the landscape (Allen-Diaz et al. 1999). Cattle replaced sheep as the primary livestock around the turn of the century as regulation, anti-Basque (shepherds) sentiment, economics, and war demands favored the production of beef and leather over mutton and wool (Allen-Diaz et al. 1999). From 1918 to 1929 around 1,800 head of cattle were permitted to graze on Sequoia National Park land, initially in support of the war effort and to relieve strain on local cattlemen caused by a drought (Stevenson 1942). Severe and long-lasting damage to several specific meadows was noted in the decades following this period of grazing, including one area where cattle trespass from a private in-holding continued to cause damage (SEKI archives 1962, 1975, 1976). During the early 1940s, after the addition of Kings Canyon as a National Park, cattlemen petitioned to resume

cattle grazing on park land. These requests were denied, citing previous grazing damage and the very high cost-to-benefit ratio for the government to allow such activity (SEKI archives 1942). However, existing permits within the Roaring River District of Kings Canyon National Park were grandfathered and grazing was allowed to continue for 20 years after the Park was designated in 1940 (Riegelhuth 1962).

Cattle have not been allowed to graze in Sequoia-Kings Canyon National Park (SEKI) since 1929 in the Sequoia portion of the Park, and since 1960 in the Kings Canyon portion. However, public lands adjacent to the Park are currently leased for the purpose of cattle grazing, including those managed by California State Lands, Bureau of Land Management, and US Forest Service. At present, approximately 40,000 cattle graze the pastures of public lands in the Sierra Nevada each summer, representing a fraction of the unknown, but extremely large numbers of cattle present at the end of the 19<sup>th</sup> century. Private lands adjacent to SEKI are also used for cattle grazing. Cattle occasionally trespass into the Park from bordering public and private lands causing resource damage. Because the damage caused by grazing can be severe and persistent, prevention of cattle trespass is important for the management and protection of Park resources.

### ***Regional Impacts***

The effects of grazing on Sierra Nevada ecosystems are generally harmful, although the degree of damage depends on the level of grazing, the type of animal, and the physical and biological properties of the landscape. Studies reporting the impacts of grazing generally do not provide a quantitative measure of grazing intensity (Allen-Diaz et al. 1999), making it difficult to draw broad conclusions. However, a summary of grazing effects is presented below.

*Terrestrial Impacts.* Impacts to terrestrial ecosystems by livestock grazing have direct effects on the soil, plants, and hydrology of these landscapes. Influences on hydrology are discussed in the subsequent Aquatic section. The most obvious and direct impact of grazing animals is on the plants that they consume. Full-grown adult cattle can weigh 700 kg (~1500 lbs) and consume about 2% of their body weight (14 kg, 30 lbs) of dry forage per day (Alberta Ag-Info Centre 2003). Grasses and sedges (graminoids) are preferred forage for cattle, comprising ~70% of their summer diet (Taylor et al. 1980). The removal of graminoid cover can allow the invasion of grasslands and meadows by less palatable species such as lodgepole pine (Norman and Taylor 2005). This can lead to a conversion of vegetation type. The grazing of grass and sedge cover can have profound and cascading effects on the physical environment as well. Consumption of standing crop and trampling of above- and below-ground plant parts can expose soil to erosive forces. The grazing impacts of soil compaction and vegetation damage combine to reduce water infiltration (thus increasing runoff) and soil-root cohesion, making soil loss and channel formation more likely (Thurow et al. 1988). Channel formation accelerates soil erosion and can lower the adjacent water table, which can cause vegetation changes and further soil erosion (Belsky et al. 1999). Many of these ecosystem responses to grazing are thought to be non-linear, exhibiting threshold dynamics where rapid, self-perpetuating degradation occurs with relatively small increases in perturbation and cannot be reversed by simply removing grazing (Sarr 2002, Friedel 1991, van de Koppel et al. 1997). Throughout the Western US these persistent grazing impacts have manifested as massive vegetation shifts (from grasslands to shrublands, from perennial grasses to annuals), soil erosion and channel incision, and degradation of aquatic habitats (Fleischner 1994).

*Aquatic Impacts.* Grazing can have multiple detrimental impacts on water quality. Cattle can serve as a direct source of water pathogens, particularly if calves, which shed more pathogens, are present (Allen-Diaz et al. 1999). A study of Sierra Nevada wilderness lakes and streams by Derlet and Carlson (2006) found all 15 sites below cattle grazed areas contained *E. coli*. Similarly, a five-year study of Sierra Nevada wilderness lakes and streams found that sites below cattle grazed areas had drastically higher occurrence of coliform bacteria, as compared to sites without domestic grazing (Derlet et al. 2008). Additionally, riparian grazing may remove shading vegetation and increase the width to depth ratio of the stream, causing negative consequences for aquatic organisms. However, this assumption is complex, and the impacts of cattle on water quality depend on the local hydrology, and the tendency of the particular cows to graze near water sources (Allen-Diaz et al. 1999).

Grazing can also affect landscape hydrology by decreasing infiltration and increasing erosion. These effects may be less severe or insignificant with lighter grazing, however, the level of impact is site specific, making optimum grazing levels difficult to determine. In addition to grazing intensity, the effects of cattle on local hydrology depend on vegetation and soil texture (Allen-Diaz et al. 1999).

*Wildlife Impacts.* In addition to terrain impacts, presence of cattle in the Sierra Nevada can also influence wildlife. In most cases, the effects of grazing on native fauna are negative. Allen-Diaz et al. (1999) summarized several examples, citing numerous sources. For example, when cattle graze on willows in riparian and meadow habitat they disrupt the nesting of the willow flycatcher (*Empidonax traillii*). Furthermore, management for cattle promotes cowbirds, which can outcompete rare native birds by increasing the availability of grain for forage. Cattle also negatively impact native ungulates by posing as competition for meadow forage and other sources of nutriment. In the presence of cattle, mule deer must increase their home range and foraging time to obtain adequate resources. A study in Idaho showed that bighorn sheep avoid cattle (Bissonette and Steinkamp 1966), although this has not been documented in the Sierra Nevada. Finally, cattle decrease habitat suitability for Golden trout by damaging undercut banks and riparian vegetation (Matthews 1996).

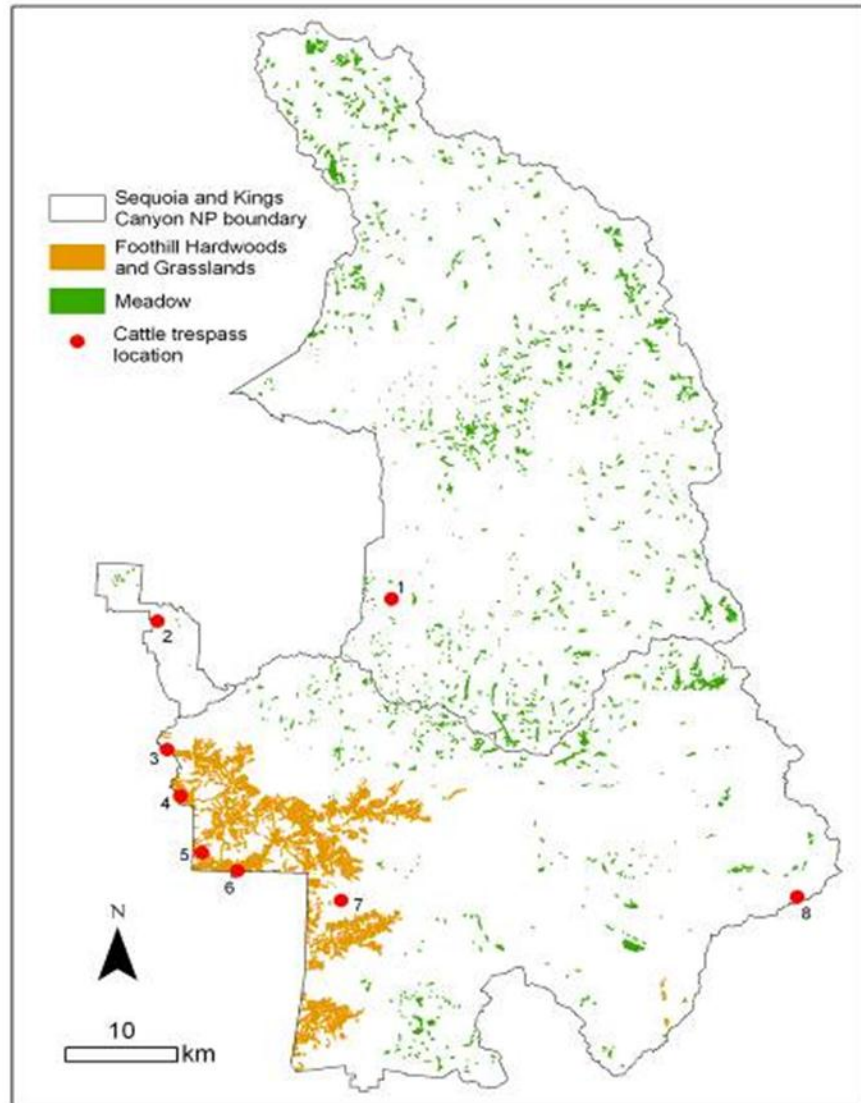
*Augmenting Invasive species.* Another risk of grazing is the distribution of invasive species. Seeds of many exotic plants remain viable after passing through cattle (e.g. Blackshaw and Rode 1991, Lyon et al. 1992, Nishida et al. 1998). The presence of *Trifolium repens* near Oriole Lake Grove Road was attributed to cattle grazing, which was observed in the area (Gerlach et al. 2003). Once invasive species establish, they can be very persistent. Keeley et al. (2003) found that a century after cessation of grazing in the Ash Mountain area there remained a similar number and percent cover of exotic species, as compared to adjacent BLM lands that had been continually grazed. Cattle can also serve as a means of dispersal for the New Zealand mud snail, which may adhere to the feet or legs of cattle using riparian areas (Proctor et al. 2007).

### ***Sequoia and Kings Canyon***

*Park impacts.* Sequoia-Kings Canyon National Park does not currently lease land for cattle grazing, however, proximity to ranches and allotments has resulted in past and current cattle trespass across park boundaries. Significant damage to grasslands, meadows, wetlands and riparian areas due to cattle trespass has been documented (Werner 1986). Most cattle trespass occurs in the southwest, low-elevation, portions of the Park where abundant hardwood, grassland, and riparian habitat abuts against grazed BLM, State of California, and private land (See locations 3-7 listed below and noted on Figure D4.1). One other area of recurring trespass is the Redwood Mountain and Canyon area, on a thin strip of Park separating National Forest land (location 2). There are also two isolated reports of trespass into a high elevation meadow (location 1, Williams Meadow, ~2,400 m) and an alpine region above treeline (location 8, Siberian Outpost, ~3,200 m). These locations are identified on Figure D4.1, and summarized below.

1. Williams Meadow. This meadow was grazed until 1960, at which point a large erosion gully had formed. This meadow was second in priority for grazing-damage repair efforts conducted by the Soil and Moisture Conservation Crew (Riegelhuth 1962). Cattle were seen trespassing in this meadow in the summer of 1997 (SEKI archives 1997).
2. Redwood Mountain. Cattle are regularly observed trespassing and grazing in this area, which includes a Giant Sequoia grove (Werner pers. comm.).
3. North Fork Kaweah, near Burnt Point. In 1986, an estimated 62 ha area was observed to have significant equid impacts and some evidence of cattle trespass (Werner 1986). This area received fencing in an attempt to prevent cattle trespass from adjacent BLM land.
4. North Fork Kaweah, near Yucca Creek. In 1986, an estimated 231 ha were impacted by trespassing cattle. This area was also fenced in an attempt to keep cattle from crossing the river to the Park side (Werner 1986).
5. Near Shepard Peak. 132 ha of land were estimated to be affected by cattle trespassing into an area that is used by the Park for its own packstock (horses and mules) grazing.
6. Near Ash Mountain. An estimated 15 ha of land were impacted by cattle trespass.
7. Oriole Lake. Cattle were observed in the Park in two consecutive years, 1996-7, and their point of entry is not well known (SEKI archives 1997).
8. Siberian Pass. An area over 3,000 m in elevation adjacent to the large Whitney grazing allotments in the Golden Trout Wilderness, Inyo National Forest. Cattle, called the Budweiser cattle because they were likely owned by Anheuser-Busch, were observed in the Park in 1995 and 1997. Impacts to aquatic resources in this area might negatively influence the rare and protected Volcano Creek Golden Trout (Sierra Club 1996).



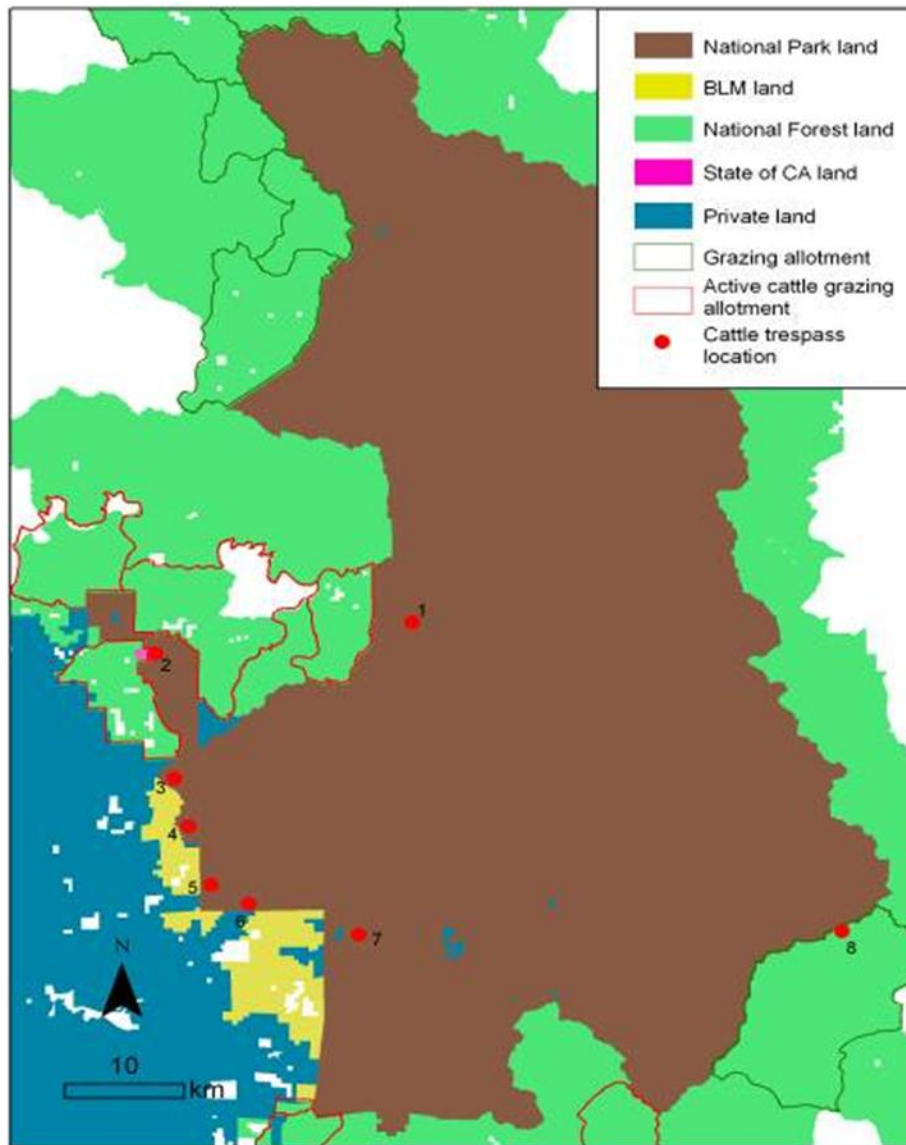


**Figure D4. 1.** The distribution of preferred forage vegetation (grasses and sedges) for cattle and locations of known cattle trespass within Sequoia and Kings Canyon National Parks.

### **Management**

*Past management practices.* Current and prior management of cattle trespass has involved establishing regulations to prohibit trespass, enforcing those rules when they are violated via warnings and fines, and physically preventing or deterring trespass using infrastructure and personnel.

Fences to prevent cattle incursion have been constructed along the Park boundary in piecemeal fashion as money and necessity arose. Maintaining and patrolling these fences is difficult and time consuming, and they often fall into disrepair (Werner 1986). It is unclear how effective the fences are at preventing cattle trespass, but it is evident in places that cattle breach damaged fences to reach the Park. In addition to erecting physical barriers to cattle, Park Rangers issue warnings and citations to ranchers whose cattle are found on Park land. In some cases the Park arranges to have the cattle moved out of the Park boundaries.



**Figure D4.2.** The ownership and management status of lands bordering Sequoia-Kings Canyon National Park and the known points of cattle trespass. Numbered trespass points are described in detail in Park Impacts. Grazing allotment data are from National Forests.

*Current risks.* The perimeter of Sequoia-Kings Canyon National Park is 481 km long. Along this perimeter the Park borders land under different ownership and management, and the threat of cattle trespass into the Park is largely dependent on the activities along these borders. The US Forest Service is the most significant neighbor of the Park, managing 382 km (80% of total) of boundary land. In the southwest of the Park, where significant trespass has occurred in the past, 32 km (7%) of Park perimeter are shared with the Bureau of Land Management. The Park borders scattered parcels of private land, including some in-holdings, along 64 km (13%) of shared boundaries, mostly in the low-elevation southwest where significant trespass has occurred. Only 3 km of Park boundary are shared with State of California lands, about 0.5% of the total Park perimeter.

Twelve grazing allotments on National Forest land border the Park. One of these (the Whitney allotment on the southeast side of the Park, where the Budweiser cattle were grazed) is currently listed as vacant. Six are listed as active cattle allotments, all on the west and south sides of the Park. Figure D4.2 shows the management status of land surrounding the Sequoia-Kings Canyon National Parks in relation to the known points of cattle trespass. In Sequoia and Kings Canyon National Parks, graminoids are concentrated in the western low-elevation oak grasslands (~8,900 ha, 2.5% of Park area) and in montane and alpine meadows (~5,500 ha, 1.6% of Park area). Figure D4.1 shows the distribution of these two vegetation types within the Park. Based on past observations of cattle trespass frequency, and the proximity of much of the Foothill Hardwoods and Grasslands vegetation to the southwest boundary, it is reasonable to expect that, as long as cattle continue to graze land adjacent to the Park, there will be a threat of trespass. Sequoia-Kings Canyon Park contains large areas of Blue Oak woodland with National Park protection status. Much of the rest of the remaining Blue Oak woodland within California is grazed or under threat of development (Davis and Stoms 1996).

Although meadows are scattered throughout the Park, they are a relatively uncommon land type, and grazing impacts can occur rapidly and be long-lasting (Sumner 1947). Where meadow damage could impact protected species, such as Giant Sequoia and Golden Trout, extra care to protect individual meadows may be warranted. The risk of cattle trespassing into the Park and causing resource degradation will be present as long as private ranch land and grazing allotments on other Public Lands abut Sequoia-Kings Canyon National Park.

### ***Cattle Trespass Stewardship***

*Critical Data Gap.* Much is known about the impacts that cattle have on ecosystems, and cattle trespass and its damage have been noted in Sequoia-Kings Canyon National Parks. The potential for future trespass is high given that grazing occurs at many points along the border of the Park. Observations of trespass and its impacts are currently done in a haphazard fashion, with no consistent protocol or methodology, making it difficult to determine the scope of the problem, or trends through time. A regular survey of likely trespass locations could generate a useful dataset that could be used to assess the size and trend of cattle trespass into the Park.

*Climate Change.* In the past, appeals to graze cattle on Park land have used two primary arguments: war time supply requirements, and extreme need of forage due to drought. While it can be assumed that modern wars will not create a huge demand for beef cattle, it is certain that periods of drought will return, and with them the increased possibility of ‘accidental’ grazing incursions in the park. Climate model predictions for the Sierra Nevada include warmer temperatures with a greater proportion of precipitation as rain (Murphy and Stine 2004), potentially increasing the need for and reducing the supply of irrigation water. This trend may exacerbate drought conditions for ranchers that rely on irrigated pastures, increasing the likelihood that they will request access to Park pastures, or allow their cattle to trespass.

### ***Summary/Recommendations***

Given that Sequoia-Kings Canyon National Park has limited resources to devote to managing cattle trespass, we recommend the following actions, listed in order of estimated highest benefit-for-cost first:

- a) Reaffirm and update the Park-wide protocol for identifying and reporting cattle trespass along with the procedure for notifying adjacent landowners or managers and levying fines.
- b) Contact landowners and grazing permittees adjacent to the Park and reestablish a dialog about cattle trespass. Include in this discussion a description of cattle impacts, recommendations for preventing trespass, incentives for self-reporting of trespass and provision of help to remove trespass cattle, and a notification of the Park's intent to pursue each incident with a fine, with repeat offenses receiving higher fines.
- c) Establish a regular monitoring program where areas of known or likely trespass (especially along the SW border) are surveyed for cattle trespass or its evidence. This program may be easiest and most cost effective to implement using camera traps along trails, abandoned roads, and riparian corridors where cattle have been known to enter the Park.
- d) Erect fences or physical barriers where cooperation from ranchers, fines, and other methods fail to protect Park resources from cattle trespass.

## D.5. Bullfrog *Rana catesbeiana*

### **Abstract**

The bullfrog (*Rana catesbeiana*) is an introduced invasive frog in the western United States. Bullfrogs often compete with and consume native turtles, frogs, salamanders and other species. They are difficult to eradicate and will likely benefit from climate warming.

- **Degree of Invasion:** minimal
- **Degree of Invasiveness:** Significant
- **Impacts:** predation on native insects and aquatic vertebrate larvae
- **Control Status:** No on-going action
- **Management recommendations:** Increased monitoring for locations and spread.

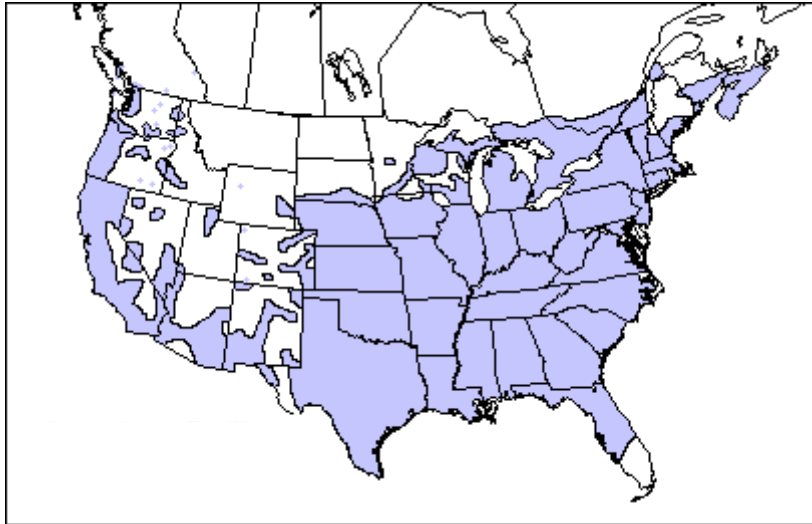
### **Species background**

*Life history.* American bullfrogs (*Rana catesbeiana*; hereafter, bullfrogs) are native to the eastern and central United States and are introduced and established in California as well as in many other parts of the world. They are considered an invasive species because of their negative impacts on aquatic ecosystems (Hecnar & M'Closkey 1997). They have been shown to prey upon native amphibians, small turtles, and songbirds (Kiesecker & Blaustein 1997, Kupferberg 1997). They can also carry diseases that can harm native amphibians such as *chytridiomycosis* and ranavirus (Sredl et al. 2000). They can breed in flowing or still water and lay hundreds of eggs which hatch very quickly (Bury & Whelan 1984). They generally take two seasons to metamorphose, and thus they are generally not found in non-permanent waters.

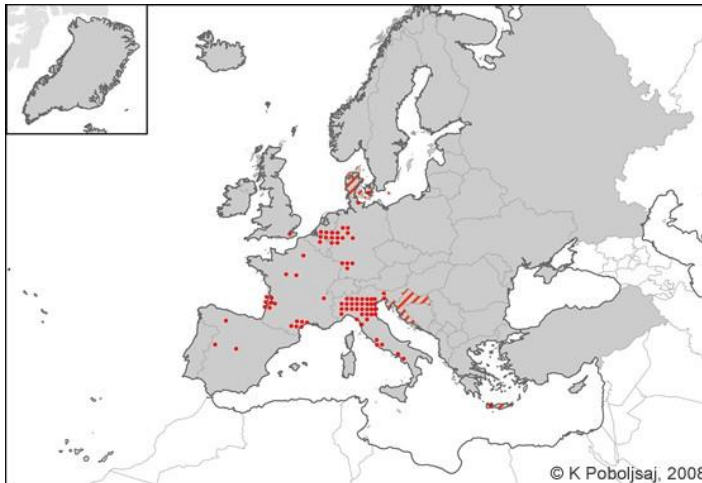
*Taxonomy.* Although this species is officially known as *Rana catesbeiana*, it is now often referred to as *Lithobates catesbeianus*. The genus *Lithobates* is often classified as a sub-genus of *Rana*, which consists of approximately 90 taxa.

### **Distribution**

Bullfrogs are native to eastern North America and are now common throughout most of North America and much of South Europe, Southeast Asia, Japan, and Hawaii, and have also been introduced to South America and the Caribbean Islands (Nussbaum et al. 1983, Bury & Whelan 1984, Stumpel 1992, Thiesmeier et al. 1994, Kupferberg 1997; Figures D5.1 & D5.2). Many of these introductions were for culinary appeal.



**Figure D5.1** Distribution of *R. catesbeiana* in the United States.



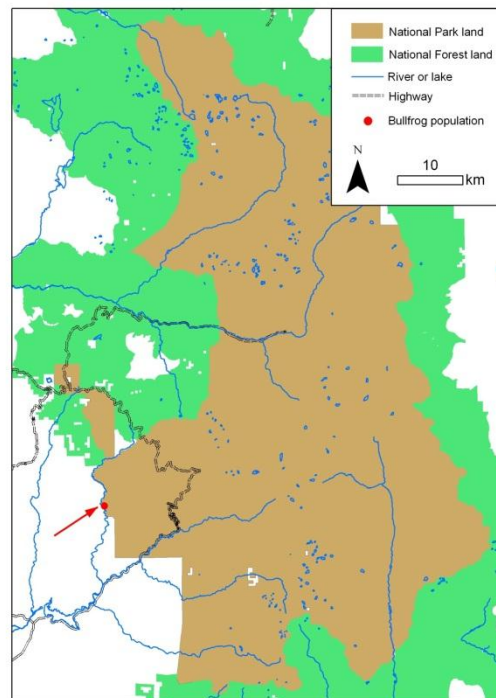
**Figure D5.2** Confirmed presence of *R. catesbeiana* in Europe.

*Population trends.* Bullfrogs are continually locally introduced, often by households and schools releasing metamorphosed bullfrogs after purchasing tadpoles online to observe metamorphosis (e.g. from the website [www.pondmegastore.com](http://www.pondmegastore.com)). Bullfrogs are increasing in most areas due to introductions and an increasing density of suitable habitats (e.g., golf course, farm ponds, drainage ponds) (Willis et al. 1956). Bullfrogs, however, are declining in some areas due to harvesting and habitat destruction and degradation (Mossman et al. 1998).

### ***Sequoia and Kings Canyon National Parks distribution***

Bullfrogs occupy very little of Sequoia and Kings Canyon National Parks, and these parks represent a very small portion of the frog's range. The initial introduction of bullfrogs into the Three Rivers area, just outside the parks, occurred in 1928, as documented by Walter Fry (1936). Bullfrogs have been found periodically in the North Fork of the Kaweah River (Figure D5.3) on the west side of the park, but not as a resident population. During a large flow event in December 1997, most or all the bullfrogs were washed out of North Fork of the Kaweah, and have been

slowly returning since (H. Werner pers. comm.). This low gradient river has many pools 1 to 4 m deep, making it particularly well-suited for bullfrogs (Lanoo 2005). There are not many similar creeks in the parks. The population in the North Fork of the Kaweah is most likely replenished from a persistent population that lives downstream from the parks' boundary in the lower North Fork and mainstem of the Kaweah River. The bullfrog population size and distribution along the North Fork of the Kaweah is not known. However, based on limited observations, it appears to be fairly dynamic: fluctuating in size and possibly location through time. Bullfrogs have returned to the North Fork following the 1997 flood, and their tadpoles have been seen being eaten by garter snakes (H. Werner pers. comm.).



**Figure D5.3.** Location of the known bullfrog population in Sequoia and Kings Canyon National Parks.

### ***Threats***

Bullfrogs are large and aggressive aquatic predators that can dramatically alter local populations of aquatic insects and displace native frogs from ponds. For example, bullfrogs may eat hatchlings of the native western pond turtle. Bullfrogs have also been implicated in the decline of native ranid frogs and garter snakes (Rosen and Schwalbe 1995).

The summer following the large flood of December 1997, when most or all bullfrogs were washed out of the North Fork of the Kaweah River, a larger than normal cohort of western pond turtles were born (H. Werner, pers. comm.). In addition to being excellent habitat for the western pond turtle, the North Fork of the Kaweah is probably good habitat for California red legged frog, a threatened species. However, there are no reports of red-legged frogs in the North Fork of the Kaweah.

***Data needs***

An accurate assessment of the size and location of the bullfrog population is needed. Because notable shifts in population size have been observed, survey efforts should be conducted in several different years, with particular attention paid to post-flood effects on population size and distribution. Continued monitoring of western pond turtles in relation to the number of bullfrogs would be helpful in determining if bullfrogs are having an impact on turtle populations. In addition, stomach content analysis of any captured bullfrogs would be beneficial in assessing their food source in the parks.

***Summary / recommendations***

It is recommended to continue monitoring water bodies for bullfrogs, especially those which have had bullfrogs in the past. If bullfrogs do become a problem, it is recommended to attempt to remove them at night with a net and headlamp or a .22 caliber gun. As it is unlikely that bullfrogs will invade additional areas of the parks due to habitat requirements, it is recommended to concentrate on the North Fork of the Kaweah River. In addition it would be valuable to initiate a dialog with landowners and managers along the mainstem of the Kaweah outside of the park regarding the removal or reduction of what is presumably the source bullfrog population for this area. Bullfrogs from this habitat would be the most likely source for reinvasion up the North Fork. Bullfrogs are also known to do better in pools warmer than those currently found in the park. If climate change increases water temperatures in the parks, it could allow bullfrogs to expand their distribution.



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